New York State
Department of Environmental Conservation

Division of Environmental Remediation
625 Broadway, Albany, New York 12233-7016

ONONDAGA LAKE

BASELINE ECOLOGICAL RISK ASSESSMENT

Syracuse, New York

NYSDEC revision prepared by: **TAMS** AN EARTH TECH COMPANY - 655 Third Avenue, New York, NY 10017

and **YEC, Inc.** - Valley Cottage, NY

Original document prepared by: **Exponent** - Bellevue, WA

for **Honeywell** - East Syracuse, NY

Volume 1 of 2
(Text, Tables, and Figures)

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Onondaga Lake Project
Site No. 7-34-030-002
Contract Number C004365, Task Order 1

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Volume 2 of 2: Appendices (contents for appendices follow appendix cover page)

List of Figures *(figures follow text of each chapter)*

Acronyms and Abbreviations

ACJ	Amended Consent Judgment
AET	apparent effects threshold
AhR	aryl hydrocarbon receptor
AQUIRE	Aquatic Information Retrieval Database
ARAR	applicable or relevant and appropriate requirement
ATSDR	Agency for Toxic Substances and Disease Registry
AVS	acid volatile sulfide
AWQC	ambient water quality criterion
BBL	Blasland, Bouck & Lee, Inc.
BEHP	bis(2-ethylhexyl)phthalate
BERA	baseline ecological risk assessment
BSAF	biota-sediment accumulation factor
BTEX	benzene, toluene, ethylbenzene, and xylenes
CBR	critical body residue
CERCLA	Comprehensive Environmental Response Compensation and Liability Act of 1980
CFR	Code of Federal Regulations
CLP	contract laboratory program
COC	chemical of concern
COPC	chemical of potential concern
CSO	combined sewer overflow
CWA	Clean Water Act
DDT	dichlorodiphenyl trichloroethane
DO	dissolved oxygen
dw	dry weight
ECL	Environmental Conservation Law
EO	Executive Order
EPC	exposure point concentration
ER-L	effects range-low
ER-M	effects range-median
ERAGS	Ecological Risk Assessment Guidance for Superfund
ERED	Environmental Residue Effects Database
ERG	Eastern Research Group
ESA	Endangered Species Act
FCV	final chronic value
FMR	field metabolic rate
FIR	food ingestion rate
FS	feasibility study
ft	feet
FWA	Freshwater Wetlands Act

FWIA	Fish and Wildlife Impact Analysis
GB	Geddes Brook
GM –IFG	General Motors – former Inland Fisher Guide
HCB	hexachlorobenzene
HCl	hydrochloric or muriatic acid
HEAST	Health Effects Assessment Summary Tables
HHRA	human health risk assessment
Honeywell	Honeywell International Inc.
HPAH	high molecular weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
IRIS	Integrated Risk Information System
km	kilometer
LCP	Linden Chemicals and Plastics
LDC	Lakefront Development Corporation
LEL	lowest effect level
LPAH	low molecular weight polycyclic aromatic hydrocarbon
LOAEL	lowest observed adverse effect level
m	meter
Metro	Metropolitan Syracuse Sewage Treatment Plant
mgd	million gallons per day
MGP	manufactured gas plant
mi	mile
N	nitrate
NAPL	non-aqueous phase liquid
NCDC	National Climatic Data Center
NCI	National Cancer Institute
NCO	non-chironomidae/oligochaeta
NMC	Ninemile Creek
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NEC	no-effect concentration
NLM	National Library of Medicine
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
NTP	National Toxicology Program
NWI	National Wetlands Inventory
NYNHP	New York Natural Heritage Program
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
NYSDOL	New York State Department of Law
OCDDS	Onondaga County Department of Drainage and Sanitation

OCDWEP	Onondaga County Department of Water Environment Protection
OLMC	Onondaga Lake Management Conference
OME	Ontario Ministry of the Environment
ORNL	Oak Ridge National Laboratory
OU	operable unit
P	phosphorous
PAH	polycyclic aromatic hydrocarbon
PCA	principal component analysis
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PCH	polychlorinated hydrocarbons
PEC	probable effect concentration
PEL	probable effect level
PMA	percent model affinity
ppb	parts per billion
ppm	parts per million
ppt	parts per thousand
PRP	potentially responsible party
PSA	preliminary site assessment
PTE	1-phenyl-1-[4-methylphenyl]-ethane, or PhenylTolyEthane
PTI	PTI Environmental Services
PXE	1-phenyl-1-[2,4-dimethylphenyl]-ethane, or PhenylXylylEthane
QA/QC	quality assurance/quality control
RAGS	Risk Assessment Guidance for Superfund
RfD	reference dose
RI	remedial investigation
ROD	Record of Decision
RME	reasonable maximum exposure
SARA	Superfund Amendments and Reauthorization Act
SCS	Soil Conservation Service
SEC	sediment effect concentration
SEL	severe effect level
SEM	simultaneously extracted metals
SIR	sediment/soil ingestion rate
SOC	stressor of concern
SOPC	stressor of potential concern
SPDES	State Pollutant Discharge Elimination System
SQB	sediment quality benchmark
SQV	sediment quality value
SUNY	State University of New York
SUNY ESF	SUNY College of Environmental Science and Forestry

SVOC	semivolatile organic compound
TAL	Target Analyte List
TAMS	TAMS Consultants, Inc.
TBC	to-be-considered
TCDD	tetrachlorodibenzo-*p*-dioxin
TCL	Target Compound List
TEC	toxic equivalent concentration
TEF	toxicity equivalence factor
TEL	threshold effects level
TEQ	toxicity equivalence quotient
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
TOGS	Technical and Operational Guidance Series
TRV	toxicity reference value
TSCA	Toxic Substances Control Act
UCL	upper confidence limit
UF	uptake factor
UPL	upper prediction limit
UFI	Upstate Freshwater Institute
UNEP	United Nations Environment Programme
USACE	United States Army Corps of Engineers
USDOE	United States Department of Energy
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
VOC	volatile organic compound
WHO	World Health Organization
WIR	water ingestion rate
WSDE	Washington State Department of Ecology
WSS	winter stress syndrome
ww	wet weight
YOY	young-of-year

Acid-Volatile Sulfide. The sulfides, consisting mainly of hydrogen sulfide and iron sulfide, removed from sediment by cold acid extraction. AVS is a method used to predict toxicity in sediment of simultaneously extracted divalent metals including cadmium, copper, lead, nickel, and zinc.

Aquatic macrophyte. Macroscopic (visible to the naked eye) forms of vegetation in the waters of the lake.

Area Use Factor. The ratio of an organism's home range, breeding range, or feeding/foraging range to the area of contamination of the site under investigation.

Assessment Endpoint. An explicit expression of the environmental value that is to be protected.

Benthic Community. The community of organisms dwelling at the bottom of a pond, river, lake, or ocean.

Bioaccumulation. General term describing a process by which chemicals are taken up by an organism, whether directly from exposure to a contaminated medium or by consumption of food containing the chemical.

Bioconcentration. A process by which there is a net accumulation of a chemical directly from an exposure medium into an organism.

Body Burden. The concentration or total amount of a substance in a living organism.

Charophytes. A group of green algae (class Charophyceae) found primarily in freshwater that are large, structurally complex algae. They range in size from a few millimeters to over a meter in length, and consist of a complex set of branching filaments.

Chronic. Involving a stimulus that is lingering or continues for a long time; often signifies periods from several weeks to years, depending on the reproductive life cycle of the species. Can be used to define either the exposure or the response to an exposure (effect). Chronic exposures typically induce a biological response of relatively slow progress and long duration.

Chronic Response. The response of (or effect on) an organism to a chemical that is not immediately or directly lethal to the organism.

Chronic Tests. A toxicity test used to study the effects of continuous, long-term exposure of a chemical or other potentially toxic material on an organism.

Community. An assemblage of populations of different species within a specified location and time.

Dietary Accumulation. The net accumulation of a substance by an organism as a result of ingestion in the diet.

Dose. A measure of exposure. Examples include (1) the amount of a chemical ingested, (2) the amount of a chemical absorbed, and (3) the product of ambient exposure concentration and the duration of exposure.

Ecosystem. The biotic community and abiotic environment within a specified location and time, including the chemical, physical, and biological relationships among the biotic and abiotic components.

Epilimnion. The upper, warm, circulating water in a thermally stratified lake in summer.

Eutrophic. Describing a body of water (e.g., a lake) with an abundant supply of nutrients and a high rate of formation of organic matter by photosynthesis. Pollution of a lake by sewage or fertilizers renders it eutrophic (a process called eutrophication). This stimulates excessive growth of algae; the death and subsequent decomposition of these increases the biochemical oxygen demand and thus depletes the oxygen content of the lake.

Exposure Pathway. The course a chemical or physical agent takes from a source to an exposed organism. Each exposure pathway includes a source or release from a source, an exposure point, and an exposure route. If the exposure point differs from the source, transport/exposure media (i.e., air, water) also are included.

Exposure Point Concentration. The concentration of a contaminant occurring at an exposure point.

Exposure Route. The way a chemical or physical agent comes in contact with an organism (i.e., by ingestion, inhalation, or dermal contact).

False Negative. The conclusion that an event (e.g., response to a chemical) is negative when it is in fact positive.

False Positive. The conclusion that an event is positive when it is in fact negative.

Food-Chain/Food-Web Transfer. A process by which substances in the tissues of lower trophic level organisms are transferred to the higher trophic level organisms that feed on them.

Hazard Quotient (HQ). The ratio of an exposure level to a substance to a toxicity value selected for the risk assessment for that substance (e.g., LOAEL or NOAEL).

Home Range. The area to which an animal confines its activities.

Hypolimnion. The lower, cooler, non-circulating water in a thermally stratified lake in summer.

Littoral. Designating or occurring in the marginal shallow water zone of a lake.

Lowest Observable Adverse Effect Level (LOAEL). The lowest level of a contaminant evaluated in a toxicity test or biological field survey that has a statistically significant adverse effect on the exposed organisms compared with unexposed organisms in a control or reference site.

Measurement Endpoint. A measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint.

No Observed Adverse Effect Level (NOAEL). The highest level of a contaminant evaluated in a toxicity test or biological field survey that causes no statistically significant difference in effect compared with the control or a reference site.

Oncolite. Irregularly rounded, calcareous nodules that range in size from 0.5 to 30 cm and are not attached to substrates.

Plankton. Minute organisms that drift with the currents in seas and lakes. Plankton includes many microscopic animals (zooplankton) and plants (phytoplankton).

Probable Effect Concentrations (PECs). Sediment quality values established as the concentrations of individual chemicals above which adverse effects in sediments are expected to frequently occur.

Sediment Effect Concentrations (SECs). Concentrations of individual contaminants in sediments below which toxicity is rarely observed and above which toxicity is frequently observed.

Species. A group of organisms that actually or potentially interbreed and are reproductively isolated from all other such groups; a taxonomic grouping of morphologically similar individuals; the category below genus.

Taxa Richness. The total number of individual taxa in a sample. The term taxa instead of species is used, as the organisms in this study are not always identified to the species level.

Thermocline. A steep temperature gradient that exists in the middle zone (the metalimnion) of a lake and gives rise to thermally induced vertical stratification of the water. The metalimnion lies between the relatively warm epilimnion above and the cold hypolimnion below.

Toxicity Test. The means by which the toxicity of a chemical or other test material is determined. A toxicity test is used to measure the degree of response produced by exposure to a specific level of stimulus (or concentration of chemical) compared with an unexposed control.

Trophic Level. A functional classification of taxa within a community that is based on feeding relationships (e.g., aquatic and terrestrial plants make up the first trophic level, and herbivores make up the second).

Type I Error. Rejection of a true null hypothesis. The percentage of stations predicted to have effects (i.e., based on exceedance of one or more of the sediment effect concentrations) that actually had no observed effects based on the chironomid survival results.

Type II Error. Acceptance of a false null hypothesis. The percentage of stations predicted to have no effects (i.e., based on lack of exceedance of any of the sediment effect concentrations) that actually had observed effects based the chironomid survival results.

EXECUTIVE SUMMARY

This document presents the New York State Department of Environmental Conservation (NYSDEC)/TAMS Consultants, Inc. (TAMS) rewrite of Honeywell International Inc.'s (Honeywell; formerly AlliedSignal) revised baseline ecological risk assessment (BERA) report. A draft BERA report was submitted to NYSDEC by Honeywell in May 1998. Based on its review and that of the US Environmental Protection Agency (USEPA), NYSDEC and the New York State Department of Law (NYSDOL) disapproved the draft document and provided comments to Honeywell in March 1999. After completing additional sampling in 1999 and 2000, Honeywell submitted a revised BERA report in April 2001. This revised report was similarly disapproved by NYSDEC and NYSDOL in July 2001. The reasons for disapproval are outlined in the determination accompanying this BERA.

For the purposes of this report, the Onondaga Lake site includes the following areas:

- The entire lake, including all pelagic and littoral areas.
- The mouths of all tributaries to the lake, including Ley Creek, Onondaga Creek, Harbor Brook, the East Flume, Tributary 5A, Ninemile Creek, Sawmill Creek, and Bloody Brook.
- The area from the lake outlet to the water sampling location in the outlet (Station W12), approximately 650 feet (ft) (200 meters [m]) downstream of the lake near the New York State Thruway bridge.
- Two of the New York State-regulated wetlands contiguous to the lake (Wetlands SYW-6 and SYW-12).

In addition to the investigations performed at the above-listed areas, ongoing or completed investigations conducted separately by Honeywell, NYSDEC, and others at hazardous waste sites and areas of concern near Onondaga Lake are discussed in the BERA.

The implementation of the BERA follows the Superfund risk assessment process specified by USEPA (1997a) to evaluate the likelihood that adverse ecological effects are occurring or may occur as a result of exposure to one or more contaminants or stressors (see text box below). The specifications of NYSDEC (1994a), particularly those specifications that are not identified by USEPA (1997a, 1998), have been incorporated into this BERA, so that the relevant New York State guidance was accommodated within the structure recommended by USEPA.

The first seven steps of the Superfund ecological risk assessment process were completed from 1990 through the present, inclusive of this report, and the final step will be determined by the NYSDEC and USEPA, with the assistance of NYSDOH and NYSDOL, during the feasibility study (FS) and Record of Decision (ROD) process.

The Eight Steps of the Superfund Ecological Risk Assessment Process

1.) Screening-level problem formulation and ecological effects evaluation.

2.) Screening-level preliminary exposure estimate and risk calculation.

3.) Baseline risk assessment problem formulation.

4.) Study design and data quality objectives.

5.) Field verification of sampling design.

6.) Site investigation and analysis of exposure and effects.

7.) Risk characterization.

8.) Risk management.

1. Honeywell History Associated with Onondaga Lake

Honeywell's predecessor companies have operated manufacturing facilities in Solvay, New York, since 1884. The location was primarily chosen due to its natural deposits of salt and limestone. The Solvay Process Company, founded in 1881, used the ammonia soda (Solvay) process to produce soda ash. Honeywell (as AlliedSignal) subsequently expanded the operation to three locations which shall be referred to in this BERA as the Main Plant, the Willis Avenue Plant and the Bridge Street Plant, collectively known as the Syracuse Works. The Main Plant manufactured soda ash and related products from 1884 to 1986 and benzene, toluene, xylenes, and naphthalene from 1917 to 1970. The Willis Avenue plant manufactured chlorinated benzenes and chlor-alkali products from 1918 to 1977. Chlor-alkali production by the mercury cell electrolytic process began in approximately 1947 at the Willis Avenue plant. The Bridge Street plant produced chlor-alkali products and hydrogen peroxide using the mercury cell electrolytic process starting in 1953. This plant was sold to Linden Chemicals and Plastics (LCP) of New York in 1979, which operated it until 1988.

An important feature of the waste management at the Syracuse Works was the use of approximately 2,000 acres of wastebeds located in Solvay (Solvay Wastebeds) to dispose of waste from the manufacture of soda ash. Honeywell disposed of Solvay wastes in these wastebeds and organic wastes in the Semet Residue Ponds in Wastebed A; organic wastes were also disposed of in Wastebed B near Harbor Brook. In addition, Honeywell disposed of large quantities of combined Solvay wastes and mercury and organic wastes directly into the lake through the East Flume. Further discussion of these and other sources is provided in Chapter 2 and Appendix G of this BERA and in the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

2. Screening-Level Problem Formulation and Screening

Initial screening-level problem formulation for Onondaga Lake was largely completed during preparation of the Onondaga Lake RI/FS Work Plan (PTI, 1991). As part of the work plan, a conceptual site model was developed, preliminary chemicals of potential concern/stressors of potential concern (COPCs/SOPCs) and representative ecological receptors were identified, assessment and measurement endpoints were defined, the objectives of the BERA were formulated, and a study design was developed to collect the data needed to satisfy the BERA objectives. Although initial problem formulation for the work plan was largely completed in 1991, several elements of the screening-level problem formulation have been refined since that time, based on information collected during the 1992 and 1999/2000 RI field investigations, or by using information collected by other parties, such as NYSDEC. The RI field investigations conducted by Honeywell in 1992 and 1999/2000 and by NYSDEC in 2002 cover the site investigation portions of Steps 4 to 6 of the USEPA Superfund ecological risk assessment process.

The preliminary conceptual site model for the Onondaga Lake BERA, which was retained with minor revisions as the site conceptual model for the BERA, is presented in Figure ES-1. The conceptual site model identifies primary and secondary sources, potential pathways, major contaminants/stressor groups, potential exposure routes and receptors, and effects to be initially evaluated as part of the BERA. Animals and plants are directly exposed to contaminants and stressors primarily from contaminated sediments and lake water and animals are indirectly exposed through ingestion of food (e.g., prey) containing contaminants.

3. Contaminants/Stressors of Concern

Numerous potentially toxic chemicals, including mercury, cadmium, chromium, copper, lead, nickel, zinc, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), benzene, toluene, ethylbenzene, xylenes (BTEX), chlorinated benzenes, and dioxins/furans, were detected at elevated concentrations in various lake media. For each complete exposure pathway, route, and chemical, a screening ecotoxicity value was selected to establish contaminant exposure levels that represent conservative thresholds for adverse ecological effects. COCs selected for water, surface sediment, surface soil, plants, fish, and wildlife receptors are presented in Tables ES-1 and ES-2.

Stressors identified in Superfund guidance are referred to as chemical contaminants in this BERA, whereas non-Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) stressors, such as chloride, depleted dissolved oxygen (DO), and reduced water transparency, are referred to as stressors. Only chemicals covered under CERCLA Section 40 CFR Part 302.4, which lists the CERCLA hazardous substances, were included in the COC selection. The exception to this is ammonia which is listed as a hazardous substance in the CFR, but is treated as an SOC in this BERA since it is associated with discharges from the Metropolitan Syracuse Sewage Treatment Plant (Metro), as well as various Honeywell sites, and is a nutrient. The major groups of stressors in Onondaga Lake, including nutrients (i.e., nitrite, phosphorus, sulfide), calcite, salinity, ammonia, depleted DO, and reduced water transparency, were retained for further examination in the BERA.

4. Assessment Endpoints

Assessment endpoints are explicit expressions of the actual environmental values that are to be protected and focus a risk assessment on particular components of the ecosystem that could be adversely affected due to contaminants and stressors at the site. Assessment endpoints are often expressed in terms of populations or communities. Because mercury and some of the other COCs, such as PCBs and polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (PCDD/PCDFs), at Onondaga Lake are known to bioaccumulate, an emphasis was also placed on indirect exposure at various levels of the food chain to address COC-related risks at higher trophic levels. In addition, assessment endpoints were also selected for communities that may have been affected by stressors. The 13 assessment endpoints that were selected for Onondaga Lake are:

- Sustainability (i.e., survival, growth, and reproduction) of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of a benthic invertebrate community that can serve as a food source for local fish and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of local fish populations.
- Sustainability (i.e., survival, growth, and reproduction) of local amphibian and reptile populations.
- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local benthivorous waterfowl populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local carnivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous (aquatic and terrestrial insect phases) mammal populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous mammal populations.

5. Measurement Endpoints

Measurement endpoints provide the actual values used to evaluate each assessment endpoint. Measurement endpoints generally include measured or modeled concentrations of chemicals and stressors in water, sediment, fish, birds, and/or mammals, laboratory toxicity studies, and field observations. Measurement endpoints in relation to their respective assessment endpoints were phrased in relation to respective risk questions contained in the BERA. Each assessment endpoint in this BERA had a minimum of two measurement endpoints that were used as lines of evidence. Measurement endpoints identified for the Onondaga Lake BERA include:

- Community structure (aquatic macrophytes, phytoplankton, zooplankton, fish, amphibians and reptiles) as compared to reference communities.

- Laboratory (greenhouse studies) and field experiments measuring macrophyte growth and survival.

- Laboratory toxicity studies measuring macroinvertebrate, growth, survival, and reproduction.

- Benthic community indices, such as richness, abundance, diversity, and biomass.

- Observed effects on fish foraging and nesting.

- Observed fish abnormalities.

- Measured total COC body burdens in fish to determine exceedance of effect-level thresholds based on toxicity reference values (TRVs).

- Laboratory toxicity studies examining effects of lake water on amphibian embryos.

- Modeled total COC body burdens in wildlife receptors to determine exceedance of effect-level thresholds based on TRVs.

- Exceedance of criteria for concentrations of COCs/SOCs in lake water that are protective of aquatic organisms, fish, and wildlife.

- Exceedance of guidelines for concentrations of COCs/SOCs in sediments that are protective of aquatic life.

- Exceedance of guidelines for concentrations of COCs/SOCs in soils that are protective of plant life.

- Field observations.

6. Ecological Receptors

The risks to the environment were evaluated for receptors that were selected to be representative of various communities, feeding preferences, predatory levels, and aquatic and wetland habitats. Individual assessment endpoints were evaluated with a minimum of one "model" (receptor) species. The following receptors were selected for the Onondaga Lake BERA:

- Aquatic macrophyte community.

- Phytoplankton community.

- Zooplankton community.

- Terrestrial plant community.

- Benthic invertebrate community.

- Fish: bluegill (*Lepomis macrochirus*); carp (*Cyprinus carpio*); channel catfish (*Ictalurus punctatus*); gizzard shad (*Dorosoma cepedianum*); largemouth bass (*Micropterus salmoides*); smallmouth bass (*Micropterus dolomieui*); walleye (*Stizostedion vitreum*); and white perch (*Morone americana*).

- Amphibian and reptile communities.

- Insectivorous birds: tree swallow (*Tachycineta bicolor*).

- Benthivorous waterfowl: mallard (*Anas platyrhynchos*).

- Piscivorous birds: belted kingfisher (*Ceryle alcyon*); great blue heron (*Ardea herodias*); and osprey (*Pandion haliaetus*).

- Carnivorous birds: red-tailed hawk (*Buteo jamaicensis*).

- Insectivorous mammals: little brown bat (*Myotis lucifugus*) – aquatic invertebrates; short-tailed shrew (*Blarina brevicauda*) – terrestrial invertebrates.

- Piscivorous mammals: mink (*Mustela vison*) and river otter (*Lutra canadensis*).

7. Exposure Assessment

The exposure assessment describes complete exposure pathways and exposure parameters. The contaminants and ecological components of the Onondaga Lake ecosystem were temporally and spatially characterized to obtain an exposure profile. The distribution of chemicals and stressors in each medium (i.e., lake water, surface sediments, wetland surface soil, dredge spoil surface soils, plankton, macroinvertebrates, and fish) to which ecological receptors may be exposed was examined and exposure point concentrations (EPCs) were calculated. Biota uptake and food-web exposure models were developed.

Receptor parameters, such as body weight, prey ingestion rate, home range, etc., were used in the food-web models to calculate COC dietary doses for wildlife. Exposure parameters were obtained from USEPA references, the scientific literature, and directly from researchers. The resulting exposure profiles for each receptor quantified the spatial and temporal patterns of exposure as they relate to the assessment endpoints and risk questions.

8. Effects Assessment

The effects assessment describes the methods used to characterize effects on aquatic and terrestrial organisms due to exposure to chemicals and stressors. Chemical exposure was evaluated using measures of toxicological effects (TRVs) that provide a basis for estimating whether the chemical exposure at a site is likely to result in adverse ecological effects. Exposure to stressors was evaluated using available literature, concentrating on studies specific to Onondaga Lake when possible.

For chemical exposure, TRVs were selected based on lowest observed adverse effects levels (LOAELs) and/or no observed adverse effects levels (NOAELs) from laboratory and/or field-based studies reported in the scientific literature. These TRVs examine the effects of COCs on the survival, growth, and reproduction of fish and wildlife species in Onondaga Lake. Reproductive effects (e.g., egg maturation, egg hatchability, and survival of juveniles) were generally the most sensitive exposure endpoints and were selected when available and appropriate.

Site-specific sediment effect concentrations (SECs) using toxicity and chemistry data were derived to allow assessment of whether the sediment chemical concentrations found at various stations in the lake would result in adverse biological effects. Five site-specific SECs were developed for Onondaga Lake using the apparent effects threshold (AET) approach and calculation of effects range-low (ER-L), effects range-median (ER-M), probable effect level (PEL), and threshold effects level (TEL) concentrations. These SECs were then used to derive a consensus-based probable effect concentration (PEC) for use in determining areas of the lake bottom that potentially pose a risk to the benthic community.

9. Risk Characterization

Risk characterization integrates the exposure and effects assessments and examines the likelihood of adverse ecological effects occurring as a result of exposure to chemicals and/or stressors. The Onondaga Lake BERA employed a strength-of-evidence approach, using several lines of evidence to evaluate each assessment endpoint.

Toxicological risks were estimated by comparing the results of the exposure assessment (measured or modeled concentrations of chemicals in receptors of concern) to the TRVs developed in the effects assessment, resulting in a ratio of these two numbers, called a hazard quotient (HQ). HQs equal to or greater than 1.0 ($HQ \geq 1$) are typically considered to indicate potential risk to ecological receptors; for example, with reduced or impaired reproduction or recruitment. The HQs provide insight into the potential for adverse effects upon individual animals in the local population resulting from chemical exposure. If an HQ suggests that effects are not expected to occur for the average individual, then they are probably insignificant at the population level. However, if an HQ indicates that risks are present for the average individual, then risks may be present for the local population.

Other measurement endpoints, such as field observations and toxicity studies, were evaluated in conjunction with toxicological risks on a receptor-specific basis. Use of several lines of evidence resulted in the following risk characterizations for each assessment endpoint.

9.1 Sustainability (i.e., Survival, Growth, and Reproduction) of an Aquatic Macrophyte Community That Can Serve as a Shelter and Food Source for Local Invertebrates, Fish, and Wildlife

Sustainability of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife was assessed using three lines of evidence. The first was comparison of the Onondaga Lake macrophyte community to reference location communities. The second was to evaluate growth and survival of macrophytes in Onondaga Lake using field and laboratory studies. The third was a qualitative evaluation of lake conditions relative to NYSDEC narrative water quality standards (6 NYCRR Part 703.2). All three measurement endpoints indicate that the macrophyte community of Onondaga Lake has been adversely affected by the input of chemicals and stressors into the lake. These impacts may affect animals that use the macrophytes in Onondaga Lake for food and shelter.

9.2 Sustainability (i.e., Survival, Growth, and Reproduction) of a Phytoplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

Sustainability of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife was assessed using two lines of evidence. The first was field observations of the Onondaga Lake phytoplankton community and the second was a qualitative evaluation of NYSDEC narrative water quality standards. Both measurement endpoints indicate that the phytoplankton community has been impacted by chemicals and/or stressors in lake water. Mercury has been shown to bioaccumulate in phytoplankton in Onondaga Lake and may be passed on to higher trophic levels feeding on phytoplankton in Onondaga Lake. Stressors have been shown to influence the abundance and distribution of phytoplankton species.

9.3 Sustainability (i.e., Survival, Growth, and Reproduction) of a Zooplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

Sustainability of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife was assessed using three lines of evidence. The first was field observations of the Onondaga Lake zooplankton community. The second was to compare surface water concentrations to water quality criteria developed for the protection of aquatic life. The third was a comparison of contaminant concentrations in sediment to NYSDEC and/or USEPA sediment guidelines. All three of these lines of evidence indicate that the zooplankton community of Onondaga Lake has been impacted by high levels of chemicals and/or stressors in lake water. In particular, high levels of salinity and mercury appear to have influenced community structure and abundance. Although the zooplankton community has been impacted by lake conditions, it still serves as a food source for local invertebrates, fish, and wildlife, and as such passes bioaccumulative contaminants (e.g., mercury) through the food chain.

9.4 Sustainability (i.e., Survival, Growth, and Reproduction) of a Terrestrial Plant Community That Can Serve as a Shelter and Food Source for Local Invertebrates and Wildlife

Sustainability of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife was assessed using two lines of evidence. The first was field observations of the Onondaga Lake terrestrial plant community. Only obvious effects, such as the sparse vegetation found on the wastebeds, can be directly attributed to activities at Honeywell facilities (i.e., disposal of Solvay and other industrial wastes). The second was to compare surface soil concentrations to plant toxicity values. Comparisons of soil chemical concentrations to plant toxicity values indicate that high levels of contaminants, in particular chromium and mercury, may adversely affect the plant community and subsequently local invertebrates and wildlife that live or forage in local habitats. These results suggest the potential for adverse effects on plants via exposure to COCs in soils at all four wetland areas and the dredge spoils area.

9.5 Sustainability (i.e., Survival, Growth, and Reproduction) of a Benthic Invertebrate Community That Can Serve as a Food Source for Local Fish and Wildlife

The potential effect of COCs and SOCs on the benthic community in Onondaga Lake was evaluated using the following four lines of evidence: exceedance of water quality criteria, benthic community metrics analysis, sediment toxicity testing, and sediment chemistry through the derivation of site-specific PECs.

Concentrations of chemicals in Onondaga Lake water were found to exceed surface water criteria in certain areas of the lake. There were more exceedances of surface water criteria in the tributaries to Onondaga Lake than in the lake itself. In addition, stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, and phosphorus, generally exceeded guidelines (when available) or background levels. A qualitative evaluation of NYSDEC narrative water quality standards indicated that those standards were also exceeded.

The benthic invertebrate community metrics analyzed in the BERA included: taxa richness, dominance, abundance of indicator species, species diversity, and percent model affinity (PMA). The analysis of these metrics showed that many of the benthic invertebrates communities living in the littoral zone (less than 5 m depth) in Onondaga Lake and the mouths of its tributaries have been impacted to some degree. The majority of moderately and severely impacted stations were located between Tributary 5A and Ley Creek, with the most severely impacted stations located between Tributary 5A and Onondaga Creek.

Short-term (10-day) and long-term (40/42-day) bulk sediment toxicity tests were performed for this BERA using sediments collected from all lake environs. The results of the sediment toxicity tests confirmed that some Onondaga Lake sediments are toxic to benthic invertebrates and may increase mortality and reduce the growth and fecundity of these organisms. The most toxic sediments are found in the nearshore zone in the southern part of the lake between Tributary 5A and Ley Creek.

Five SECs (i.e., calculation of AET, ER-L, ER-M, PEL, and TEL values) were derived to allow site-specific assessment of whether the sediment chemical concentrations found at various Onondaga Lake stations would result in adverse biological effects. These SECs were then used to derive a consensus-based PEC (i.e, the contaminant concentration above which adverse effects are expected to frequently occur) to determine areas of the lake bottom that pose some degree of risk to the benthic community. The PECs were derived as the geometric mean of the five site-specific SECs and are presented in Table ES-3.

Using the consensus PECs, measured surface sediment concentrations exceed the values at many locations throughout Onondaga Lake. Only 14 of approximately 200 locations sampled in 1992 and 2000 do not have at least one compound exceeding an HQ of 1.0 (i.e., sediment concentration less than the PEC). Many of the ratios of measured sediment concentrations to PECs exceed 10, or even 100, between Tributary 5A and Ley Creek. In addition, these sediment locations have the highest number of compounds – between 11 and over 30 compounds per sample – that exceed their PECs in a sample.

Based on the above, all four lines of evidence suggest an adverse effect from COCs and SOCs on the benthic invertebrate populations in Onondaga Lake, particularly in the southern part of the lake from Tributary 5A to Ley Creek. Based on these analyses it can also be concluded that local fish and wildlife populations using the benthic invertebrate community as a food source in turn are impacted.

9.6 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Fish Populations

The sustainability of local fish populations was assessed using six lines of evidence. The first was to examine the fish community structure as compared to similar lakes and historic accounts of Onondaga Lake (prior to industrial activities) in relation to the health of local fish populations. The second was to look for potential effects of chemicals/stressors on fish foraging and nesting. The third was to compare visual abnormalities (e.g., tumors, lesions) in Onondaga Lake fish to fish from other lakes. The fourth was to compare measured water column concentrations to water quality criteria for the protection of aquatic life, including NYSDEC narrative standards. The fifth was to compare measured sediment concentrations to guidelines for the protection of aquatic life for benthic-dwelling species of fish. The sixth and final line of evidence was to compare measured concentrations of chemicals in fish representing various feeding strategies and trophic levels to TRVs.

Risks to fish from chemicals were evaluated on a species-specific basis using measured body burdens for eight fish species representing the Onondaga Lake fish community (Table ES-4). A limited number of chemicals (e.g., methylmercury) were analyzed in some species (e.g., gizzard shad and largemouth bass). Therefore, actual risks from chemicals in lake water may be greater for these species than calculated. HQs greater than 1.0 were calculated for the following chemicals (by species):

- Bluegill – arsenic, chromium, endrin, mercury, selenium, vanadium, and zinc.
- Carp – arsenic, chromium, dioxin/furans, endrin, mercury, total PCBs, selenium, vanadium, and zinc.
- Catfish – chromium, endrin, methylmercury, mercury, total PCBs, selenium, vanadium, and zinc.
- Gizzard shad – methylmercury.
- Largemouth bass – methylmercury and dioxins/furans.
- Smallmouth bass – arsenic, chromium, mercury, methylmercury, total PCBs, selenium, vanadium, and zinc.
- Walleye – chromium, mercury, methylmercury, and total PCBs.
- White perch – chromium, mercury, methylmercury, selenium, and total PCBs.

Five of the six lines of evidence evaluated suggest adverse effects from COCs on the Onondaga Lake fish community and the remaining line of evidence, incidence of visual abnormalities, was inconclusive. This strength-of-evidence approach indicates that local fish populations are adversely affected by the chemicals and stressors present in Onondaga Lake.

9.7 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Amphibian and Reptile Populations

Sustainability of local amphibian and reptile populations was assessed using three lines of evidence. The first was to conduct a field survey of local amphibian and reptile populations around Onondaga Lake. The second was to compare measured water column concentrations to water quality criteria for the protection of aquatic life, including NYSDEC narrative standards. The third and final line of evidence was laboratory studies examining the effects of Onondaga Lake water on amphibian embryos. All three lines of evidence strongly indicate that amphibian and reptile populations have been adversely affected by chemicals and/or stressors found in Onondaga Lake water.

9.8 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Insectivorous Bird Populations

Sustainability of local insectivorous bird populations was assessed using three lines of evidence. The first was modeling dietary doses of chemicals. The second was to compare measured water column concentrations to water quality criteria for the protection of wildlife. The third line of evidence was field-based observation. The first two lines of evidence suggested that insectivorous birds have been adversely affected to some degree by chemicals found in Onondaga Lake and taken up by the aquatic phases (e.g., egg, larvae) of invertebrates. Mercury HQs were up to an order-of-magnitude greater than 1.0 and PAH HQs were up to two orders-of-magnitude greater than 1.0, with both COCs exceeding a HQ of 1.0 over the full concentration and toxicity range evaluated (Table ES-5). The third line of evidence, field observations, was inconclusive.

9.9 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Benthivorous Waterfowl Populations

Sustainability of local waterfowl populations was assessed using three lines of evidence. The first was modeling dietary doses of chemicals. The second was to compare measured water column concentrations to water quality criteria for the protection of wildlife. The third line of evidence was field-based observation. The first two lines of evidence suggested that waterfowl have been adversely affected to some degree by chemicals found in Onondaga Lake via exposure to contaminated water and food sources. Mercury HQs were up to an order-of-magnitude greater than 1.0 and PAH HQs were up to two orders-of-magnitude greater than 1.0, with both COCs exceeding a HQ of 1.0 over the full concentration and toxicity range evaluated (Table ES-5). The third line of evidence, field observations, was inconclusive.

9.10 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Piscivorous Bird Populations

Sustainability of local piscivorous bird populations was assessed using three lines of evidence. The first was modeling dietary doses of chemicals. The second was to compare measured water column concentrations to water quality criteria for the protection of wildlife. The third line of evidence was field-based observation. The first two lines of evidence suggested that piscivorous birds have been adversely affected to some degree by chemicals found in Onondaga Lake, and by mercury in particular. Mercury HQs were greater than 1.0 for the full point estimate range of risk for all three piscivorous receptor species and were over an order-of-magnitude greater than the NOAELs (Table ES-5). The third line of evidence, field observations, was inconclusive.

9.11 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Carnivorous Bird Populations

Sustainability of local carnivorous bird populations was assessed using two lines of evidence. The first was modeling dietary doses of chemicals and the second was field-based observation. Modeled dietary doses suggested that carnivorous birds have been adversely affected to some degree by chemicals found in Onondaga Lake, and by total PAHs in particular, for which HQs were greater than 1.0 for the full point estimate range of risk (Table ES-5). The second line of evidence, field observations, was inconclusive.

9.12 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Insectivorous (Aquatic and Terrestrial Insect Phases) Mammal Populations

Sustainability of local insectivorous mammal populations was assessed using three lines of evidence. The first was modeling dietary doses of chemicals. The second was to compare measured water column concentrations to water quality criteria for the protection of wildlife. The third line of evidence was field-based observation. The first two lines of evidence suggested that insectivorous mammals feeding on aquatic invertebrates have been adversely affected to some degree by chemicals found in Onondaga Lake. Methylmercury and PAHs had the highest HQs, with HQs greater than 1.0 for the full point estimate range of risk and values up to an order-of-magnitude above 1.0 (Table ES-6).

Insectivorous mammals feeding on terrestrial invertebrates in the four wetlands around Onondaga Lake may also be adversely affected by chemicals found in Onondaga Lake. Risk varied by wetland area, with SYW-19, located near the mouth of Harbor Brook, having the greatest number of COCs with HQs above 1.0 (Table ES-7). In the wetland areas, risks from exposure to methylmercury for the full point estimate range of risk in all four wetlands were up to two orders-of-magnitude above 1.0. Risks from exposure to total PAHs, hexachlorobenzene, and dioxins/furans were up to three orders-of-magnitude above 1.0. Risks to insectivorous mammals in the dredge spoils soils were primarily due to exposure to hexachlorobenzene. The third line of evidence, field observations, was inconclusive.

9.13 Sustainability (i.e., Survival, Growth, and Reproduction) of Local Piscivorous Mammal Populations

The sustainability of local piscivorous mammal populations was assessed using three lines of evidence. The first was modeling dietary doses of chemicals. The second was to compare measured water column concentrations to water quality criteria for the protection of wildlife. The third line of evidence was field-based observation. The first two lines of evidence suggested that piscivorous mammals feeding around Onondaga Lake have been adversely affected to some degree by chemicals found in the lake, and in particular by mercury and total PCBs (Table ES-6). The third line of evidence, field observations, was inconclusive.

10. Uncertainties

To integrate the various components of the BERA, the results of the risk characterization and associated uncertainties were evaluated to assess the risk of adverse effects to Onondaga Lake receptors as a result of exposure to chemicals and stressors originating in the lake. Uncertainty exists because of data limitations (e.g., extrapolating between species for TRVs) and natural variability (e.g., fish tissue concentrations, ingestion rates). Uncertainty is an inherent component of risk assessments. Elements of uncertainty in this BERA were identified and efforts were made to minimize them. For components in which a moderate degree of uncertainty was unavoidable (e.g., sampling data), efforts were made to minimize any systematic bias associated with the data. The Onondaga Lake BERA uses various point estimates of exposure and response to develop a range of point estimates of risk (i.e., 95 percent UCL, mean, NOAEL, and LOAEL) to aid in judging the ecological significance of risks.

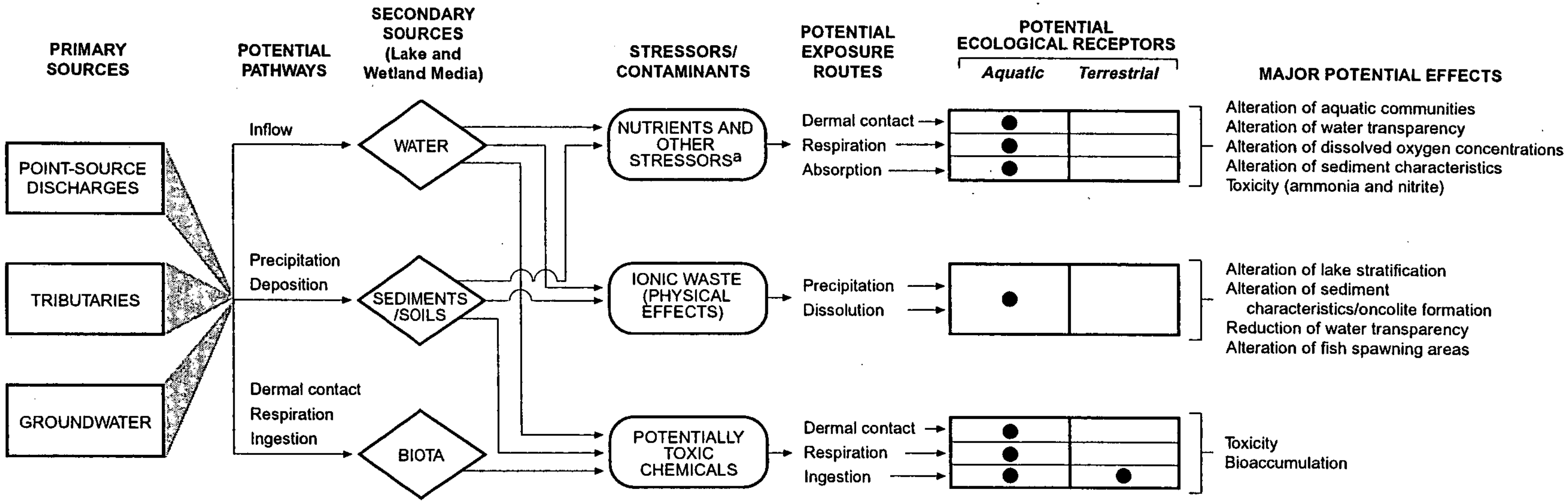
In addition to the uncertainties that are common to many risk assessments, there were several uncertainties associated with this BERA that are specific to Onondaga Lake. Uncertainties associated with factors limiting the distribution and abundance of macrophytes, the effects of calcium and oncolites on the aquatic community, the effects on the Onondaga Lake ecosystem if conditions allow the return of an oxic hypolimnion, and the effects of eutrophication on the lake ecosystem were examined and discussed in the BERA.

11. Conclusions

Multiple lines of evidence were used to evaluate major components of the Onondaga Lake ecosystem to determine if lake contamination has adversely affected plants and animals around Onondaga Lake. Almost all lines of evidence indicate that the Honeywell-related contaminants and ionic waste in Onondaga Lake have produced adverse ecological effects at all trophic levels examined.

The aquatic macrophytes in the lake have been adversely affected by lake conditions, and the resulting loss of macrophyte habitat that formerly provided valuable feeding and nursery areas has undoubtedly affected the aquatic invertebrates and vertebrates living in Onondaga Lake. In addition to general habitat loss, there has been bioaccumulation of mercury and possibly other chemicals in most organisms serving as a food source in the lake, including phytoplankton, zooplankton, benthic invertebrates, and fish. Exceedances of site-specific sediment PECs suggest adverse effects to benthic invertebrates will frequently occur (Ingersoll et al., 2000) in most areas of the lake. The greatest number and magnitude of exceedances were found in areas in the southern portion of the lake and near Ninemile Creek (see Chapter 10, Figure 10-3).

Comparisons of measured tissue concentrations and modeled doses of chemicals to TRVs show exceedances of HQs for site-related chemicals throughout the range of the point estimates of risk. Many of the contaminants in the lake are persistent and therefore, the risks associated with these contaminants are unlikely to decrease significantly in the absence of remediation. On the basis of these comparisons, it has been determined through this BERA that all receptors of concern are at risk. Contaminants and stressors in the lake have either impacted or potentially impacted every trophic level and feeding preference examined in this BERA.



[a] Other stressors include calcite, salinity, ammonia, dissolved oxygen, transparency, wave scour, and non-native species

Source: Modified from Exponent, 2001b

Figure ES-1. Conceptual Site Model for the Baseline Ecological Risk Assessment for Onondaga Lake

Table ES-1. Contaminants of Concern Selected for Onondaga Lake Media

Chemical	Water	Sediment	Soil	Plants	Fish
Metals					
Antimony		•	•		•
Arsenic		•	•	•	•
Barium	•		•		
Cadmium		•	•	•	
Chromium		•	•	•	•
Copper	•	•	•	•	
Iron			•		
Lead	•	•	•	•	
Manganese	•	•	•		
Mercury/Methylmercury	•	•	•	•	•
Nickel		•	•	•	
Selenium		•	•	•	•
Silver		•	•	•	
Thallium			•	•	
Vanadium		•	•	•	•
Zinc	•	•	•	•	•
Cyanide	•		•		
Volatile Organic Compounds					
Benzene		•	•		
Chlorobenzene	•	•	•		
Dichlorobenzenes (Sum)	•	•	•		
Ethylbenzene		•			
Toluene		•			
Trichlorobenzenes (Sum)	•	•	•		
Xylene isomers		•			
Semivolatile Organic Compounds					
Bis(2-ethylhexyl)phthalate	•				
Dibenzofuran		•			
Hexachlorobenzene		•	•		
Phenol		•	•		
Polycyclic aromatic hydrocarbon (total)		•	•		
Pesticides/Polychlorinated Biphenyls					
Aldrin			•		
Chlordane isomers		•	•		
DDT and metabolites		•	•		•
Dieldrin		•	•		
Endrin					•
Hexachlorocyclohexanes			•		
Heptachlor and heptachlor epoxide		•			
Polychlorinated biphenyls (total)		•	•		•
Dioxins/Furans					
Total dioxins/furans		•			•

Note: • – Contaminants of concern assessed in the BERA for the specific media listed.
DDT – dichlorodiphenyltrichloroethane

Table ES-2. Contaminants of Concern for Wildlife Species Evaluated in the Onondaga Lake BERA

Chemicals of Concern	Tree Swallow	Mallard	Belted Kingfisher	Great Blue Heron	Osprey	Red-Tailed Hawk	Little Brown Bat	Short-Tailed Shrew	Mink	River Otter
Metals										
Antimony							•	•		
Arsenic	•						•	•	•	•
Barium	•	•					•	•		
Cadmium	•	•					•	•		
Chromium	•	•	•	•	•	•	•	•	•	•
Cobalt	•	•					•			
Copper	•	•					•			
Lead	•		•			•	•	•		
Manganese							•			
Mercury/Methylmercury	•	•	•	•	•	•	•	•	•	•
Nickel	•	•					•			
Selenium	•		•	•	•		•	•	•	•
Thallium	•						•	•		
Vanadium	•	•					•	•	•	•
Zinc	•	•	•	•	•		•	•		
Volatile Organic Compounds										
Dichlorobenzenes (total)	•	•								
Trichlorobenzenes (total)	•	•					•	•		
Xylenes (total)	•	•					•			
Semivolatile Organic Compounds										
Bis(2-ethylhexyl)phthalate	•									
Hexachlorobenzene							•	•	•	
Polycyclic aromatic hydrocarbon (total)	•	•	•	•		•	•	•	•	•
Pesticides/Polychlorinated Biphenyls										
Chlordanes								•		
DDT and metabolites	•		•	•	•	•			•	
Dieldrin							•	•	•	•
Endrin			•							
Hexachlorocyclohexanes			•	•	•					
Polychlorinated biphenyls (total)	•	•	•	•	•		•	•	•	•
Dioxins/Furans										
Dioxins/furans (TEQ)	•	•	•		•	•	•	•	•	•

Notes: • – Contaminants of concern (COC) assessed in the BERA for the specific receptor listed.
DDT – dichlorodiphenyltrichloroethane
TEQ – toxicity equivalent

Table ES-3. Comparison of Various Site-Specific Sediment Effect Concentrations and Probable Effect Concentrations for Onondaga Lake, 1992 Data[ab]

	AET	ER-L	ER-M	TEL	PEL	PEC
Metals (mg/kg)						
Antimony	NC	3.1	3.1	4	4.3	3.6
Arsenic	4.3	0.9	4.4	1.3	3.6	2.4
Cadmium	8.6	0.9	2.1	1.4	3.1	2.4
Chromium	195	18	48	29	67	50
Copper	84	12	41	19	48	33
Lead	116	9.7	57	13	58	35
Manganese	445	197	280	231	295	278
Total mercury	13	0.5	2.8	1.0	2.8	2.2
Nickel	50	5.2	21	8.4	26	16
Selenium	0.9	0.4	0.6	0.4	0.7	0.6
Silver	2.7	0.8	1.2	0.9	1.4	1.3
Vanadium	12	2.7	6.0	3.4	8.3	5.6
Zinc	218	38	95	57	12	88
Organic Compounds						
BTEX Compounds (μg/kg)						
Benzene	5,300	27	42	42.4	299	150
Ethylbenzene	13	142	657	206	657	176
Toluene	443	13	28	16	50	42
Xylenes	606	153	1,640	367	997	561
Chlorinated Benzenes (μg/kg)						
Chlorobenzene	10,000	64	580	48	799	428
Dichlorobenzenes	1,373	21.5	773	44	765	239
Trichlorobenzenes	287	186	930	209	482	347
Hexachlorobenzene	28	7.2	28	8.9	24	16
Polychlorinated Biphenyls (μg/kg)						
Aroclor 1016	90	99	135	104	135	111
Aroclor 1248	470	82	300	99	307	204
Aroclor 1254	77	69	83	74	80	76
Aroclor 1260	240	80	240	115	221	164
Total PCBs	710	136	400	151	382	295
PAH Compounds (μg/kg)						
Naphthalene	2,100	340	1,400	471	1,380	917
Acenaphthene	1,700	469	1,200	478	1,030	861
Fluorene	3,500	55	305	66.9	327	264
Phenanthrene	16,000	92	480	135	491	543
Anthracene	4,400	33	210	49.6	249	207
Fluoranthene	26,000	140	1,400	483	2,482	1,436
Pyrene	NC	114	650	238	795	344
Benz[a]anthracene	NC	61	415	118	451	192
Chrysene	NC	100	440	172	541	253
Benzo[b]fluoranthene	1,100	63	240	81	253	908
Benzo[a]pyrene	NC	63	210	98	355	146
Indeno[1,2,3-cd]pyrene	NC	59	370	102	503	183
Dibenz[a,h]anthracene	730	49	180	67.7	218	157
Benzo[ghi]perylene	2,700	228	1,300	307	1,170	780
Acenapthylene	3,000	507	1,850	673	1,970	1,301
Benzo[k]fluoranthene	1,100	63	240	81	253	203
Dibenzofuran	NC	340	340	295	561	372

Table ES-3. (cont.)

	AET	ER-L	ER-M	TEL	PEL	PEC
Other SVOCs (μg/kg)						
Phenol	45	45	45	45	45	45
Pesticides (μg/kg)						
DDT and Metobolites	16	47	47	24	27	30
Chlordane	NC	NC	NC	5.1	5.1	5.1

Notes:

[a] All concentrations in dry weight

[b]Maps of exceedances of ER-L, ER-M, TEL, PEL and PEC values are presented in Appendix F.

AET - apparent effects threshold
BTX - benzene, toluene, xylenes
DDT – dichlorodiphenyltrichloroethane
ER-L - effects-range low
ER-M - effects-range median
NC - value was not calculated because of an insufficient number of detected observations
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
PEC - Probable Effect Concentration
PEL - probable effect level
TEL - threshold effect level

Table ES-4. Hazard Quotients for Measured Fish Concentrations

COC	Bluegill 95%UCL HQ NOAEL	Bluegill 95%UCL HQ LOAEL	Bluegill Mean HQ NOAEL	Bluegill Mean HQ LOAEL	Gizzard Shad 95%UCL HQ NOAEL	Gizzard Shad 95%UCL HQ LOAEL	Gizzard Shad Mean HQ NOAEL	Gizzard Shad Mean HQ LOAEL
Antimony	0**	0**	0**	0**	0*	0*	0*	0*
Arsenic	1.4	0.5	0.7	0.3	0*	0*	0*	0*
Chromium	61	18	16	4.6	0*	0*	0*	0*
Mercury	5.4	1.8	2.7	0.9	0*	0*	0*	0*
Methylmercury	3.5	1.2	2.8	0.9	2.3	0.8	2.1	0.7
Selenium	15	1.5	9.2	0.9	0*	0*	0*	0*
Vanadium	29	2.9	20	2.0	0*	0*	0*	0*
Zinc	3.2	2.7	2.1	1.8	0*	0*	0*	0*
Endrin	0.2	2.3E-02	0.1	1.5E-02	0*	0*	0*	0*
DDT and metabolites	4.7E-02	9.7E-03	3.9E-02	8.0E-03	0*	0*	0*	0*
Polychlorinated biphenyls	0.5	0.1	0.3	0.1	0*	0*	0*	0*
Dioxin/furan TEQ (Fish)	0.4	0.2	0.1	0.1	0*	0*	0*	0*

Table ES-4. (cont.)

CoC	Carp 95%UCL HQ NOAEL	Carp 95%UCL HQ LOAEL	Carp Mean HQ NOAEL	Carp Mean HQ LOAEL	Catfish 95%UCL HQ NOAEL	Catfish 95%UCL HQ LOAEL	Catfish Mean HQ NOAEL	Catfish Mean HQ LOAEL
Antimony	0**	0**	0**	0**	0.4	0.2	6.3E-02	3.5E-02
Arsenic	**4.0**	**1.5**	**1.7**	0.6	0**	0**	0**	0**
Chromium	**21**	**6.2**	**7.2**	**2.1**	**5.7**	**1.7**	**3.1**	0.9
Mercury	**4.3**	**1.4**	**3.5**	**1.2**	**6.3**	**2.1**	**4.9**	**1.6**
Methylmercury	**4.8**	**1.6**	**3.9**	**1.3**	**7.8**	**2.6**	**7.1**	**2.4**
Selenium	**20**	**2.0**	**10**	**1.0**	**13**	**1.3**	**7.6**	0.8
Vanadium	**24**	**2.4**	**13**	**1.3**	**27**	**2.7**	**20**	**2.0**
Zinc	**13**	**11**	**6.1**	**5.2**	**2.2**	**1.8**	**1.2**	**1.0**
Endrin	**1.0**	0.1	0.5	0.0	0.8	0.1	0.5	0.0
DDT and metabolites	0.4	0.1	0.3	0.1	0.6	0.1	0.3	0.1
Polychlorinated biphenyls	**2.5**	0.5	**1.6**	0.3	**2.1**	0.4	**1.5**	0.3
Dioxin/furan TEQ (Fish)	**2.6**	**1.2**	**1.0**	0.5	0.6	0.3	0.4	0.2

Table ES-4. (cont.)

CoC	White Perch 95%UCL HQ NOAEL	White Perch 95%UCL HQ LOAEL	White Perch Mean HQ NOAEL	White Perch Mean HQ LOAEL	SMB 95%UCL HQ NOAEL	SMB 95%UCL HQ LOAEL	SMB Mean HQ (NOAEL)	SMB Mean HQ (LOAEL)
Antimony	0.4	0.2	0.4	0.2	0**	0**	0**	0**
Arsenic	0**	0**	0**	0**	3.6	1.4	2.4	0.9
Chromium	2.5	0.7	2.5	0.7	3.2	0.9	2.3	0.7
Mercury	7.7	2.6	7.0	2.3	7.3	2.4	7.0	2.3
Methylmercury	12	4.1	11	3.6	8.2	2.7	7.2	2.4
Selenium	7.8	0.8	7.8	0.8	10	1.0	4.8	0.5
Vanadium	0**	0**	0**	0**	20	2.0	11	1.1
Zinc	0.5	0.4	0.5	0.4	1.6	1.4	1.1	0.9
Endrin	0.1	1.4E-02	0.1	1.2E-02	0.2	1.7E-02	0.2	1.6E-02
DDT and metabolites	0.2	3.5E-02	0.1	1.3E-02	0.1	2.1E-02	0.1	1.5E-02
Polychlorinated biphenyls	1.3	0.3	1.1	0.2	1.0	0.2	0.9	0.2
Dioxin/furan TEQ (Fish)	0.5	0.3	0.4	0.2	0.5	0.2	0.3	0.1

Table ES-4. (cont.)

CoC	LMB 95%UCL HQ NOAEL	LMB 95%UCL HQ LOAEL	LMB Mean HQ NOAEL	LMB Mean HQ LOAEL	Walleye 95%UCL HQ NOAEL	Walleye 95%UCL HQ LOAEL	Walleye Mean HQ NOAEL	Walleye Mean HQ LOAEL
Antimony	NA	NA	NA	NA	0**	0**	0**	0**
Arsenic	0*	0*	0*	0*	0**	0**	0**	0**
Chromium	0*	0*	0*	0*	**3.2**	0.9	**3.2**	0.9
Mercury	**6.9**	**2.3**	**6.6**	**2.2**	**15**	**5.2**	**14**	**4.6**
Methylmercury	0*	0*	0*	0*	**18**	**6.1**	**15**	**5.1**
Selenium	0*	0*	0*	0*	0**	0**	0**	0**
Vanadium	0*	0*	0*	0*	0**	0**	0**	0**
Zinc	0*	0*	0*	0*	0**	0**	0**	0**
Endrin	0**	0**	0**	0**	0.3	2.7E-02	0.1	1.3E-02
DDT and metabolites	0.1	1.2E-02	2.9E-02	6.1E-03	0.2	3.6E-02	0.1	2.1E-02
Polychlorinated biphenyls	0.7	0.1	0.4	0.1	**2.8**	0.6	**1.5**	0.3
Dioxin/furan TEQ (Fish)	**1.4**	0.7	0.9	0.4	0*	0*	0*	0*

Notes:

* denotes not analyzed
** denotes all non-detects
Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
LMB – largemouth bass
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
SMB – smallmouth bass
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table ES-5. Hazard Quotients for Modeled Avian Exposure

COC	Tree Swallow 95% UCL HQ NOAEL	Tree Swallow 95% UCL HQ LOAEL	Tree Swallow Mean HQ NOAEL	Tree Swallow Mean HQ LOAEL	Mallard 95% UCL HQ NOAEL	Mallard 95% UCL HQ LOAEL	Mallard Mean HQ NOAEL	Mallard Mean HQ LOAEL	Belted Kingfisher 95% UCL HQ NOAEL	Belted Kingfisher 95% UCL HQ LOAEL	Belted Kingfisher Mean HQ NOAEL	Belted Kingfisher Mean HQ LOAEL
Metals												
Arsenic	0.1	4.4E-02	0.1	3.1E-02	NS	NS	NS	NS	NS	NS	NS	NS
Barium	**10**	**5.1**	**8.3**	**4.1**	**2.4**	**1.2**	**1.8**	0.9	NS	NS	NS	NS
Cadmium	**7.0**	0.5	**4.6**	0.3	**1.0**	0.1	0.7	4.7E-02	NS	NS	NS	NS
Chromium	**53**	**11**	**57**	**11**	**10**	**2.1**	**9.7**	**1.9**	0.2	3.8E-02	0.2	3.6E-02
Cobalt	NA	NA	NA	NA	NA	NA	NA	NA	NS	NS	NS	NS
Copper	0.8	0.6	0.6	0.5	0.2	0.1	0.1	0.1	NS	NS	NS	NS
Lead	**1.8**	0.2	**1.3**	0.1	NS	NS	NS	NS	0.1	1.4E-02	0.1	8.7E-03
Methylmercury	**19**	**1.9**	**11**	**1.1**	**4.3**	0.4	**2.7**	0.3	**23**	**2.3**	**20**	**2.0**
Mercury	**6.5**	**3.3**	**3.1**	**1.5**	0.9	0.4	0.7	0.3	0.7	0.3	0.6	0.3
Nickel	0.2	0.1	0.2	0.1	3.9E-02	2.8E-02	3.7E-02	2.7E-02	NS	NS	NS	NS
Selenium	**6.8**	**3.4**	**5.4**	**2.7**	NS	NS	NS	NS	3.9E-03	2.0E-03	3.1E-03	1.5E-03
Thallium	NA	NA	NA	NA	NS	NS	NS	NS	NS	NS	NS	NS
Vanadium	0.1	1.1E-02	0.1	7.9E-03	2.6E-02	2.6E-03	1.5E-02	1.5E-03	NS	NS	NS	NS
Zinc	**6.4**	0.7	**5.6**	0.6	**1.2**	0.1	**1.0**	0.1	1.0E-02	1.1E-03	8.6E-03	9.5E-04
Volatile Organic Compounds												
Xylenes	NA	NA	NA	NA	NA	NA	NA	NA	NS	NS	NS	NS
Dichlorobenzenes	**3.0**	0.3	**1.4**	0.1	**2.1**	0.2	0.3	3.3E-02	NS	NS	NS	NS
Trichlorobenzenes	NA	NA	NA	NA	NA	NA	NA	NA	NS	NS	NS	NS
Semivolatile Organic Compounds												
Bis(2-ethylhexyl)phthalate	0.7	0.1	0.6	0.1	NS	NS	NS	NS	NS	NS	NS	NS
Polycyclic aromatic hydrocarbons	**287**	**29**	**292**	**29**	**393**	**39**	**118**	**12**	**12**	**1.2**	**3.7**	0.4
Pesticides/Polychlorinated Biphenyls												
Endrin	NS	NS	NS	NS	NS	NS	NS	NS	2.9E-04	2.9E-05	2.4E-04	2.4E-05
Hexachlorocyclohexanes	NS	NS	NS	NS	NS	NS	NS	NS	2.2E-05	7.2E-06	2.0E-05	6.3E-06
DDT and metabolites	0.8	0.1	0.6	0.1	0.2	2.0E-02	0.1	1.4E-02	**19**	**1.9**	**12**	**1.2**
Polychlorinated biphenyls (PCBs)	**1.9**	0.2	**1.8**	0.2	0.4	3.9E-02	0.3	3.0E-02	**11**	**1.1**	**3.1**	0.3
Dioxins/Furans												
Dioxins/furans (TEQ) avian	**5.6**	0.6	**1.3**	0.1	**1.4**	0.1	0.3	3.1E-02	**1.8**	0.2	**1.4**	0.1

Table ES-5. (cont.)

	Great Blue Heron				Osprey				Red-tailed Hawk			
	95% UCL HQ		Mean HQ		95% UCL HQ		Mean HQ		95% UCL HQ		Mean HQ	
COC	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals												
Arsenic	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Barium	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Cadmium	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Chromium	0.1	2.7E-02	0.1	2.5E-02	0.1	2.1E-02	0.1	1.9E-02	0.2	4.7E-02	0.2	3.4E-02
Cobalt	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Copper	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Lead	NS	NS	NS	NS	NS	NS	NS	NS	0.4	4.2E-02	0.3	3.0E-02
Methylmercury	**18**	**1.8**	**15**	**1.5**	**24**	**2.4**	**20**	**2.0**	0.3	2.7E-02	0.1	7.2E-03
Mercury	0.3	0.1	0.3	0.1	0.3	0.2	0.3	0.2	0.1	7.1E-02	0.0	1.3E-02
Nickel	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Selenium	0.5	0.2	0.4	0.2	0.7	0.4	0.5	0.3	NS	NS	NS	NS
Thallium	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Vanadium	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Zinc	**1.1**	0.1	0.8	0.1	**1.6**	0.2	**1.2**	0.1	NS	NS	NS	NS
Volatile Organic Compounds												
Xylenes	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Dichlorobenzenes	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Trichlorobenzenes	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Semivolatile Organic Compounds												
Bis(2-ethylhexyl)phthalate	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Polycyclic aromatic hydrocarbons	**4.0**	0.4	**1.2**	0.1	NA	NA	NA	NA	**252**	**25**	**14**	**1.4**
Pesticides/Polychlorinated Biphenyls												
Endrin	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Hexachlorocyclohexanes	1.0E-02	3.3E-03	0.2	0.1	1.5E-02	4.8E-03	0.3	0.1	NS	NS	NS	NS
DDT and metabolites	**8.0**	0.8	**5.3**	0.5	**9.3**	0.9	**6.3**	0.6	**1.5**	0.2	0.3	0.0
Polychlorinated biphenyls (PCBs)	**2.7**	0.3	**1.4**	0.1	**2.5**	0.3	0.2	2.5E-02	NS	NS	NS	NS
Dioxins/Furans												
Dioxins/furans (TEQ) avian	NS	NS	NS	NS	0.6	0.1	0.4	4.3E-02	**9.9**	0.99	**1.0**	0.1

Notes: NA = Not Available; NS = Not selected as a COC for this receptor.
Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table ES-6. Hazard Quotients for Modeled Mammalian Exposure

COC	Little Brown Bat				Mink				River Otter			
	95% UCL HQ		Mean HQ		95% UCL HQ		Mean HQ		95% UCL HQ		Mean HQ	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals												
Arsenic	**1.1**	0.1	0.8	0.1	0.2	1.7E-02	0.1	1.1E-02	0.8	0.1	0.5	0.1
Barium	**2.1**	**1.3**	**1.7**	**1.0**	NS	NS	NS	NS	NS	NS	NS	NS
Cadmium	**4.5**	0.5	**3.0**	0.3	NS	NS	NS	NS	NS	NS	NS	NS
Chromium	**7.2**	**1.8**	**7.8**	**1.9**	0.7	0.2	0.6	0.2	0.3	0.1	0.3	0.1
Cobalt	0.4	3.9E-02	0.3	3.4E-02	NS	NS	NS	NS	NS	NS	NS	NS
Copper	**1.4**	**1.1**	**1.1**	0.9	NS	NS	NS	NS	NS	NS	NS	NS
Lead	0.1	1.2E-02	0.1	8.8E-03	NS	NS	NS	NS	NS	NS	NS	NS
Manganese	3.8E-02	1.2E-02	3.5E-02	1.1E-02	NS	NS	NS	NS	NS	NS	NS	NS
Methylmercury	**21**	**2.1**	**13**	**1.3**	**12**	**1.2**	**9.4**	0.9	**43**	**4.3**	**36**	**3.6**
Mercury	**1.3**	0.1	0.6	0.1	0.1	1.4E-02	0.1	9.9E-03	0.1	1.5E-02	0.1	1.4E-02
Nickel	0.1	0.1	0.2	8.0E-02	NS	NS	NS	NS	NS	NS	NS	NS
Selenium	0.21	0.13	0.16	0.1	0.1	0.1	0.1	7.1E-02	0.9	0.5	0.7	0.4
Thallium	0.1	7.9E-03	0.1	7.1E-03	NS	NS	NS	NS	NS	NS	NS	NS
Vanadium	**2.7**	0.3	**1.9**	0.2	0.3	2.8E-02	0.7	6.7E-02	0.8	0.1	0.6	0.1
Zinc	0.26	0.13	0.22	0.11	NS	NS	NS	NS	NS	NS	NS	NS
Volatile Organic Compounds												
Trichlorobenzenes	2.8E-02	7.8E-03	0.1	1.7E-02	NS	NS	NS	NS	NS	NS	NS	NS
Xylenes	**2.3**	**1.9**	0.5	0.4	NS	NS	NS	NS	NS	NS	NS	NS
Semivolatile Organic Compounds												
Hexachlorobenzene	**6.0**	0.6	**4.6**	0.5	**9.2**	0.9	**1.1**	0.1	NS	NS	NS	NS
Polycyclic aromatic hydrocarbons	**18**	**1.8**	**19**	**1.9**	**33**	**3.3**	**4.5**	0.4	**5.2**	0.5	**1.6**	0.2
Pesticides/Polychlorinated Biphenyls												
DDT and metabolites	NS	NS	NS	NS	1.5E-02	2.9E-03	7.5E-03	1.5E-03	**5.9**	**1.2**	**2.3**	4.5E-01
Dieldrin	0.6	0.3	0.5	0.2	0.2	0.1	0.1	0.1	0.2	7.7E-02	0.1	4.4E-02
Polychlorinated biphenyls (PCBs)	0.4	0.1	0.4	0.1	**109**	**11**	**34**	**3.4**	**130**	**13**	**69**	**6.9**
Dioxins/Furans												
Dioxins/furans (TEQ) mammalian	**11**	**1.1**	**2.9**	0.3	**42**	**4.2**	**4.9**	0.5	**2.8**	0.3	**1.5**	0.2

Notes: NA = Not Available; NS = Not selected as a COC for this receptor. Hazard quotients equal to or greater than one are outlined and bolded.

DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table ES-7. Hazard Quotients for Modeled Short-Tailed Shrew Exposure in Wetlands and Dredge Spoils Area

COC	SYW-6 95%UCL HQ NOAEL	SYW-6 95%UCL HQ LOAEL	SYW-6 Mean HQ NOAEL	SYW-6 Mean HQ LOAEL	SYW-19 95%UCL HQ NOAEL	SYW-19 95%UCL HQ LOAEL	SYW-19 Mean HQ NOAEL	SYW-19 Mean HQ LOAEL
Total Metals								
Antimony	0.4	3.6E-02	0.1	9.5E-03	0.2	1.8E-02	0.1	1.0E-02
Arsenic	2.0	0.2	1.1	0.1	2.8	0.3	2.3	0.2
Barium	0.1	0.1	0.1	0.1	0.3	0.2	0.2	0.1
Beryllium	1.9E-02	1.9E-03	1.5E-02	1.5E-03	2.6E-02	2.6E-03	2.2E-02	2.2E-03
Cadmium	11	1.1	3.5	0.4	2.5	0.3	1.6	0.2
Chromium	1.0	0.2	0.3	0.1	0.3	0.1	0.3	0.1
Lead	1.5	0.1	0.7	0.1	2.1	0.2	1.0	0.1
Methylmercury	22	2.2	19	1.9	29	2.9	27	2.7
Mercury	0.2	1.9E-02	0.1	1.1E-02	0.6	6.3E-02	0.4	4.1E-02
Nickel	3.3E-02	1.6E-02	1.5E-02	7.5E-03	2.2E-02	1.1E-02	1.6E-02	8.1E-03
Selenium	1.7	1.0	0.6	0.4	1.2	0.8	1.1	0.7
Thallium	2.6	0.3	1.4	0.1	ND	ND	ND	ND
Vanadium	2.9	0.3	1.8	0.2	1.7	0.2	1.6	0.2
Zinc	0.7	0.4	0.5	0.2	0.4	0.2	0.4	0.2
Volatile Organic Compounds								
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06	3.4	0.9	1.2	0.3
Semivolatile Organic Compounds								
Hexachlorobenzene	ND	ND	ND	ND	783	78	241	24
PAHs	213	21	47	4.7	2,565	256	794	79
Pesticides/Polychlorinated Biphenyls								
Chlordane	ND	ND	ND	ND	0.6	0.1	0.2	4.2E-02
Dieldrin	ND	ND	ND	ND	7.3	3.7	5.0	2.5
PCBs	3.9E-02	9.7E-03	2.8E-02	6.9E-03	1.8	0.5	1.4	0.4
Dioxins/Furans								
Dioxins/furans (TEQ)	15	1.5	5.9	0.6	1,706	171	681	68

Table ES-7. (cont.)

COC	SYW-12 95%UCL HQ NOAEL	SYW-12 95%UCL HQ LOAEL	SYW-12 Mean HQ NOAEL	SYW-12 Mean HQ LOAEL	SYW-10 95%UCL HQ NOAEL	SYW-10 95%UCL HQ LOAEL	SYW-10 Mean HQ NOAEL	SYW-10 Mean HQ LOAEL
Total Metals								
Antimony	0.1	9.5E-03	4.7E-02	4.7E-03	8.3E-02	8.3E-03	4.5E-02	4.5E-03
Arsenic	1.4	0.1	0.99	9.9E-02	5.3	0.5	2.3	0.2
Barium	0.1	0.1	0.1	4.6E-02	0.1	7.3E-02	0.1	4.9E-02
Beryllium	1.8E-02	1.8E-03	1.2E-02	1.2E-03	5.0E-02	5.0E-03	2.4E-02	2.4E-03
Cadmium	7.5	0.8	5.0	0.5	1.2	0.1	0.7	0.1
Chromium	0.7	0.2	0.4	0.1	0.3	0.1	0.2	4.3E-02
Lead	1.0	0.1	0.7	0.1	1.0	0.1	0.6	0.1
Methylmercury	19	1.9	19	1.9	22	2.2	20	2.0
Mercury	0.1	1.2E-02	9.4E-02	9.4E-03	0.2	1.7E-02	0.1	1.3E-02
Nickel	1.6E-02	8.1E-03	9.9E-03	4.9E-03	1.7E-02	8.6E-03	1.0E-02	5.1E-03
Selenium	0.7	0.5	0.4	0.3	1.3	0.8	0.7	0.4
Thallium	ND	ND	ND	ND	4.3	0.4	2.8	0.3
Vanadium	2.0	0.2	1.1	0.1	3.9	0.4	2.0	0.2
Zinc	0.5	0.3	0.5	0.2	0.4	0.2	0.4	0.2
Volatile Organic Compounds								
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06	5.8E-06	1.6E-06	5.6E-06	1.6E-06
Semivolatile Organic Compounds								
Hexachlorobenzene	1.8	0.2	0.5	4.9E-02	2.0	0.2	1.5	0.1
PAHs	191	19	61	6.1	155	15.5	38	3.8
Pesticides/Polychlorinated Biphenyls								
Chlordane	0.1	2.6E-02	0.1	1.3E-02	ND	ND	ND	ND
Dieldrin	1.1	0.6	0.6	0.3	ND	ND	ND	ND
PCBs	0.4	0.1	0.2	0.1	0.1	3.5E-02	5.9E-02	1.5E-02
Dioxins/Furans								
Dioxins/furans (TEQ)	NA	NA	NA	NA	4.4	0.4	3.6	0.4

Table ES-7. (cont.)

COC	Dredge Spoils 95%UCL HQ NOAEL	Dredge Spoils 95%UCL HQ LOAEL	Dredge Spoils Mean HQ NOAEL	Dredge Spoils Mean HQ LOAEL
Total Metals				
Antimony	0.1	6.5E-03	4.9E-02	4.9E-03
Arsenic	**2.7**	0.3	**1.9**	0.2
Barium	6.0E-02	3.6E-02	5.6E-02	3.3E-02
Beryllium	2.3E-02	2.3E-03	1.8E-02	1.8E-03
Cadmium	1.7E-04	1.7E-05	1.7E-04	1.7E-05
Chromium	0.2	4.6E-02	0.1	2.7E-02
Lead	0.2	1.7E-02	0.1	1.4E-02
Methylmercury	0.1	6.8E-03	5.E-02	5.E-03
Mercury	0.2	1.8E-02	9.E-02	9.E-03
Nickel	8.5E-03	4.3E-03	7.0E-03	3.5E-03
Selenium	**1.1**	0.7	0.8	0.5
Thallium	ND	ND	ND	ND
Vanadium	**3.7**	0.4	**2.4**	0.2
Zinc	0.3	0.2	0.3	0.1
Volatile Organic Compounds				
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06
Semivolatile Organic Compounds				
Hexachlorobenzene	**38**	**3.8**	**4.6**	0.5
PAHs	**9.0**	0.9	**2.0**	0.2
Pesticides/Polychlorinated Biphenyls				
Chlordane	NA	NA	NA	NA
Dieldrin	NA	NA	NA	NA
PCBs	3.4E-02	8.6E-03	1.7E-02	4.3E-03
Dioxins/Furans				
Dioxins/furans (TEQ)	0.7	0.1	0.4	4.2E-02

Notes: NA = Not available, ND = Not detected

Hazard quotients equal to or greater than one are outlined and bolded.

DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

1. INTRODUCTION

Honeywell International Inc. (Honeywell; formerly AlliedSignal) is currently conducting a comprehensive remedial investigation/feasibility study (RI/FS) of Onondaga Lake, located near Syracuse, New York (Figure 1-1). The RI/FS is being conducted under a Consent Decree with the State of New York dated January 9, 1992, as amended (Index No. 89-CV-815). The scope and details of the RI/FS were originally developed through negotiations between Honeywell, the New York State Department of Law (NYSDOL), and the New York State Department of Environmental Conservation (NYSDEC), and are specified in the Consent Decree and the approved Onondaga Lake RI/FS Work Plan (PTI, 1991), which is an appendix to the Consent Decree.

As part of the RI/FS, a draft baseline ecological risk assessment (BERA) report was submitted by Honeywell in May 1998. The BERA was reviewed by NYSDEC and the US Environmental Protection Agency (USEPA). With the concurrence of the reviewers, NYSDEC and NYSDOL disapproved this draft document and provided comments to Honeywell in March 1999. After completing additional sampling in 1999 and 2000, Honeywell submitted a revised BERA report in April 2001. This revised report was also assessed by these reviewers and, with their concurrence, NYSDEC and NYSDOL disapproved it in July 2001. The reasons for disapproval are outlined in the determination accompanying this document, which is the NYSDEC/TAMS Consultants, Inc. (TAMS) rewrite of Honeywell's revised BERA report, and it has likewise been reviewed by and has received the concurrence of NYSDOL and USEPA.

NYSDEC/TAMS obtained some information, including historical sources of contamination, in this BERA report and the accompanying RI and human health risk assessment (HHRA) (TAMS, 2002b,a), from, among other sources, reports and materials prepared by Honeywell and its consultants. While the accuracy of the information provided by Honeywell and its consultants is accepted for purposes of these reports, it must be noted that pursuant to paragraph 68 of the Consent Decree, discovery in the underlying litigation has been stayed. Consequently, the information furnished by Honeywell and its consultants, as well as information provided by third-party sources, has not been verified through the formal discovery process. The State reserves the right, consistent with and without limitation to its rights under paragraphs 33 and 34 of the Consent Decree and under state and federal law, to correct or amend any information in the BERA, RI, and HHRA if, without limitation: (a) discovery is conducted, and (b) that discovery reveals information supporting such correction or amendment.

For the purposes of this BERA, the Onondaga Lake site includes the following:

- The entire lake, including all pelagic and littoral areas.

- The mouths of all tributaries to the lake, including Ley Creek, Onondaga Creek, Harbor Brook, the East Flume, Tributary 5A, Ninemile Creek, Sawmill Creek, and Bloody Brook.

- The area from the lake outlet to the sampling location in the outlet (Station W12), approximately 650 feet (ft) (200 meters [m]) downstream of the lake near the New York State Thruway bridge.

- Wetlands SYW-6 and SYW-12.

In addition to the areas of the site listed above, this BERA includes an evaluation of limited data that were collected in Wetlands SYW-10 and SYW-19 and an upland area associated with the dredge spoils area located north of the mouth of Ninemile Creek. Ecological risk associated with Wetlands SYW-10 and SYW-19 and the dredge spoils area will be further evaluated as part of separate sites and, therefore, the ecological risk analyses associated with these areas in this BERA is considered preliminary, pending the finalization of the BERAs associated with these other sites. Specifically, Wetland SYW-10 will be further evaluated as part of the RI/FS for the Geddes Brook/Ninemile Creek site; Wetland SYW-19 will be further evaluated as part of the RI/FS for the Wastebed B/Harbor Brook site; and the dredge spoils area will be further evaluated as a separate site with its own investigation.

The perimeter of the area evaluated as part of this BERA is depicted in Figure 1-2, and the major features of Onondaga Lake are shown in a recent aerial photograph presented as Figure 1-3.

Consistent with USEPA guidance (USEPA, 1997a), a specific objective of the ecological risk assessment process is to identify and characterize the current and potential threats to the environment from a hazardous substance release. This BERA was conducted in accordance with the terms of the RI/FS Work Plan (PTI, 1991) and state and federal guidance documents, including:

- Guidelines for Ecological Risk Assessment (USEPA, 1998).

- Ecological Risk Assessment Guidance for Superfund (ERAGS): Process for Designing and Conducting Ecological Risk Assessments (USEPA, 1997a).

- Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites (USEPA, 1999a).

- Fish and Wildlife Impact Analysis (FWIA) for Inactive Hazardous Waste Sites (NYSDEC, 1994a).

In keeping with the recommendations of these agency guidance documents, this BERA focuses on hazardous substances (i.e., metals and organic compounds) identified under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended, and the National Oil and Hazardous Substances Pollution Contingency Plan (NCP). For purposes of this BERA, these CERCLA-related substances (stressor chemicals) are referred to as chemicals of concern (COCs), whereas stressors (some of which are chemicals), such as chloride, phosphorus, depleted dissolved oxygen (DO), and reduced water transparency, are referred to as stressors of concern (SOCs).

The general process and structure of the BERA are presented in Figures 1-4 and 1-5 and are consistent with USEPA guidance (USEPA, 1992a, 1997a, 1998). The BERA includes the following major components:

- **Problem formulation** – Establishes the goals and focus of the BERA. Assessment endpoints, or specific ecological values to be protected, are selected and a conceptual model is developed.

- **Exposure assessment** – Evaluates the degree to which key ecological receptors are potentially exposed to COCs and SOCs in Onondaga Lake.

- **Effects assessment** – Evaluates the degree to which exposure to COCs and SOCs in the lake may result in adverse ecological effects.

- **Risk characterization** – Estimates the degree of risk posed by COCs and SOCs in the lake and interprets the ecological significance of those risks.

The structure of the BERA has aimed to be consistent with USEPA guidelines (1997a, 1998) and follows the eight-step process specified by USEPA in ERAGS (1997a), which is presented in Figure 1-6. The equivalent of the current problem formulation component of the BERA (Steps 1 to 4) was initially conducted in 1990 to1991 by Honeywell during development of the RI/FS Work Plan (PTI, 1991), and has been revisited throughout the BERA process by Honeywell/Exponent and NYSDEC/USEPA/TAMS. The original work plan was approved in 1991 and was included as an appendix to the Consent Decree.

From 1990 to 1992, several initial studies (Step 5) were conducted by Honeywell to refine the study design described in the work plan. In 1990, a reconnaissance survey was conducted to tour the Honeywell facilities and Onondaga Lake and to develop a preliminary sampling strategy. In 1991, a pilot study was conducted to evaluate the sediment toxicity tests proposed for use in the lake and to visit candidate reference lakes (PTI, 1993a). In 1992, an initial sediment coring survey was conducted at 19 stations throughout the lake to refine the list of chemicals of potential concern (COPCs) identified in the work plan. Also in 1993, a comparative evaluation of candidate reference lakes was conducted through a review of the available literature to identify the most appropriate reference lake for use in the RI/FS (PTI, 1992c; revised by NYSDEC in 1993). As indicated in NYSDEC's comment letter on the draft BERA (Larson, pers. comm., 1999a), Otisco Lake has been designated the "reference lake" for analysis of sediment toxicity, benthic macroinvertebrates, and macrophytes.

The main site field investigation (Step 6) was conducted by Honeywell from April to December 1992 (RI Phase 1). The 1992 field investigation was subdivided into five smaller investigations corresponding to the major types of data targeted for collection. These smaller investigations are described below, along with information from each investigation that was used in the BERA:

- **Geophysical Investigation** – Information on the bathymetry of Onondaga Lake was used to stratify benthic macroinvertebrate sampling stations by water depth and to evaluate the potential for wind-induced sediment disturbance throughout the littoral zone of the lake.

- **Contaminant and Stressor Investigation** – Information on contaminants and stressor concentrations and distribution in surface sediments (0 to 2 cm) of Onondaga Lake was used to evaluate potential risks to biota in the lake.

- **Mercury and Calcite Mass Balance Investigation** – Information on mercury and calcite concentrations in the water of Onondaga Lake and its tributaries was collected. However, Honeywell did not develop acceptable models for use in the BERA (NYSDEC/TAMS, 1998b,c).

- **Ecological Effects Investigation** – Quantitative information on sediment chemistry, toxicity, and benthic macroinvertebrate communities in Onondaga Lake, as compared to a nearby reference lake (i.e., Otisco Lake), was used to evaluate potential risks to sediment-dwelling organisms in Onondaga Lake. Semi-quantitative and qualitative information on macrophyte, phytoplankton, and zooplankton communities in Onondaga Lake was combined with more quantitative information collected by other parties to evaluate potential risks to those communities in the lake.

- **Bioaccumulation Investigation** – Information on COC concentrations in sediment, surface water, benthic macroinvertebrates, and fish in Onondaga Lake was used to evaluate exposure to COCs and potential risks to fish, semiaquatic, and terrestrial receptors (i.e., benthivorous, insectivorous, and piscivorous birds and insectivorous, semi-piscivorous, and piscivorous mammals) that prey on lake biota.

A summary of the 1992 information used in the BERA is presented in Chapter 7, Table 7-1.

Following completion of the main site investigation in 1992 and submittal of the draft BERA to NYSDEC in May 1998, a supplemental field investigation was conducted by Honeywell in 1999 (Supplemental Lake Water Sampling Investigation) and 2000 (Phase 2A Investigation) to collect additional information deemed necessary by NYSDEC. Additional sampling of sediments in Wetland SYW-6 was performed by NYSDEC/TAMS in May 2002 (TAMS, 2002b). A summary of the 1999 to 2002 information used in the BERA is presented in Chapter 7, Table 7-2.

This BERA addresses the information collected in all field investigations (i.e., 1992, 1999, 2000, and 2002). Risk characterization (Step 7) has been in progress since 1994 and represents the end product of the BERA. Historical information on conditions in the lake prior to 1992 was reviewed in the RI/FS work plan and is not a subject of this BERA.

In preparing the BERA, the specifications of NYSDEC's Fish and Wildlife Impact Analysis (FWIA) process (NYSDEC, 1994a) have been incorporated. For example, terrestrial covertypes within 0.5 miles (mi) (0.8 kilometers [km]) of the lakeshore and wetlands within 2 mi (3.2 km) of the lakeshore were mapped in detail, which is not required in USEPA guidance. In this manner, relevant New York State guidance was accommodated within the structure recommended by USEPA.

Investigations at several upland sites and tributaries related to Honeywell have been proceeding concurrently with the Onondaga Lake RI/FS. Those investigations are summarized in Chapter 2 of this BERA. These upland and tributary studies evaluate the impact of Honeywell's operations on and near the upland site areas. To the extent that upland contamination is reaching or has reached Onondaga Lake, the ecological risk associated with that contamination within the boundaries of the Onondaga Lake site is evaluated as part of this BERA.

Much of the detailed information on which the BERA is based is presented in the appendices of this report. The RI/FS data collected in 1992 are presented in a series of data reports (PTI, 1993b,c,d,e). The detailed methods used to collect and analyze the RI/FS samples collected in 1992 are also presented in those data reports and the Onondaga Lake RI/FS field sampling plan (PTI, 1992a). RI/FS data collected from 1999 to 2002 are presented in the RI report (TAMS, 2002b).

The remainder of this document consists of the following 12 chapters:

- Chapter 2, Summary of Honeywell and Other Industrial Facilities and Environmental Investigations, describes Honeywell facilities and related areas near Onondaga Lake, and environmental studies conducted at those facilities.

- Chapter 3, Site Description (FWIA Step I), presents information about fish and wildlife resources near Onondaga Lake, describes fish and wildlife resource values, and identifies applicable fish and wildlife criteria.

- Chapter 4, Screening-Level Problem Formulation and Ecological Effects Evaluation (ERAGS Step 1), presents the initial screening-level steps of the ecological risk assessment, including the development of a preliminary site conceptual model and preliminary identification of COPCs and stressors of potential concern (SOPCs), ecological receptors, and assessment and measurement endpoints.

- Chapter 5, Screening-Level Exposure Estimate and Risk Calculation (ERAGS Step 2), presents the results of screening-level risk calculations used to refine the list of COPCs/SOPCs carried forward in the BERA.

- Chapter 6, Baseline Risk Assessment Problem Formulation (ERAGS Step 3), presents the baseline risk assessment problem formulation; refines COPCs and SOPCs; characterizes ecological effects of contaminants; reviews information on contaminant fate and transport, complete exposure pathways, and ecosystems potentially at risk; selects assessment endpoints and measurement endpoints; and develops a conceptual model.

- Chapter 7, Study Design (ERAGS Steps 4 and 5), describes the study design by summarizing major components of the Onondaga Lake work plan, the 1992, 1999, 2000, and 2002 field investigations, and other sources of information.

- Chapter 8, Analysis of Ecological Exposures (ERAGS Step 6), characterizes chemicals and stressors in Onondaga Lake media and presents an exposure characterization for ecological receptors.

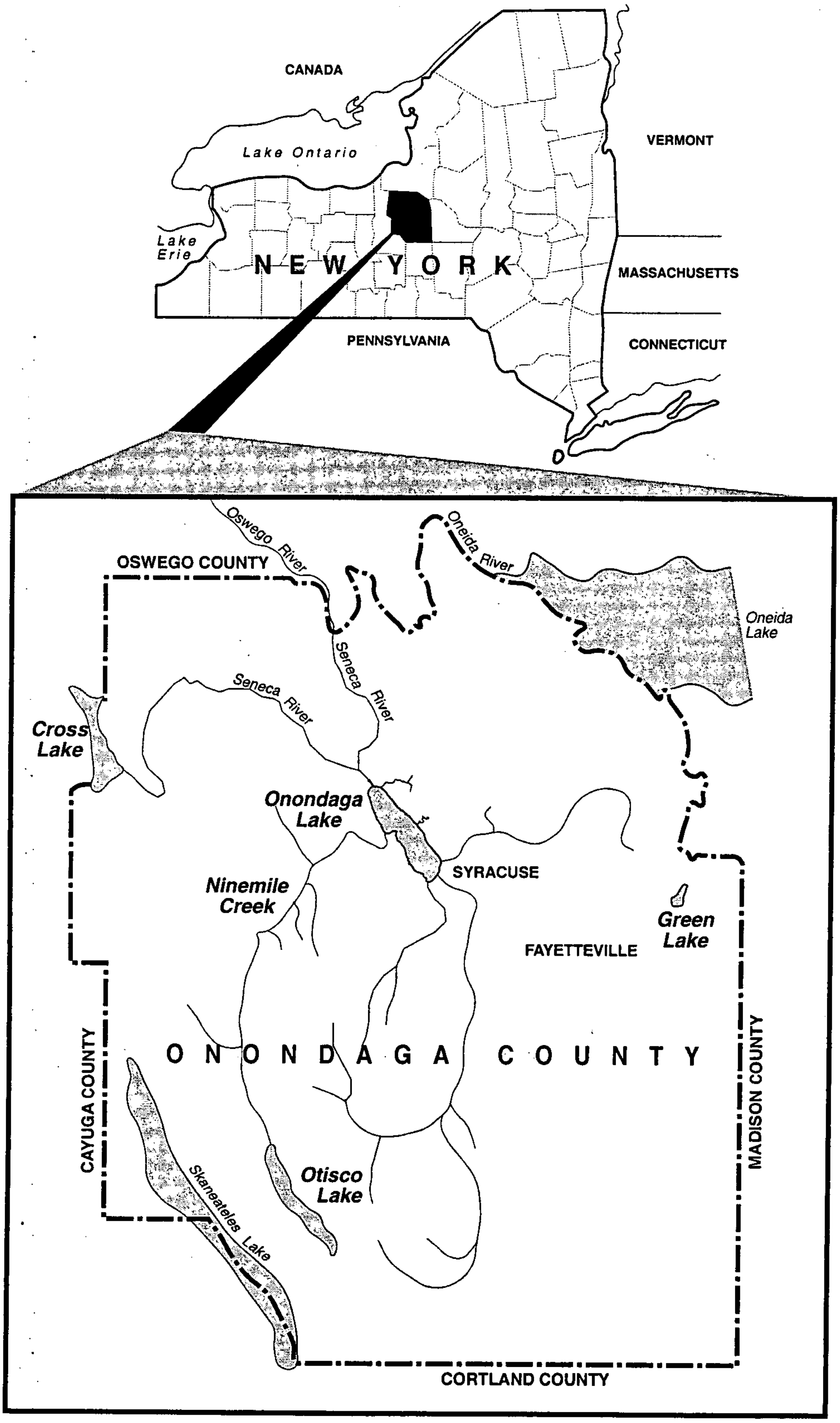
- Chapter 9, Analysis of Ecological Effects (ERAGS Step 6), presents information on effects characterization. Site-specific field investigations and observations are discussed, evidence of existing impacts based on toxicity testing is presented along with the derivation of sediment effect concentrations (SECs), and toxicity reference values (TRVs) are selected for fish and wildlife receptors.

- Chapter 10, Risk Characterization (ERAGS Step 7), integrates information on exposure and effects to estimate potential risks. Each assessment endpoint is evaluated in regard to associated measurement endpoints.

- Chapter 11, Uncertainty Analysis (ERAGS Step 7), evaluates various sources of uncertainty in the risk assessment.

- Chapter 12, Conclusions, summarizes the major findings of the ecological risk assessment.

- Chapter 13, References, presents references for all documents and personal communications cited in the main body of the report.



Source: Exponent, 2001b.

Figure 1-1. Location of Onondaga Lake

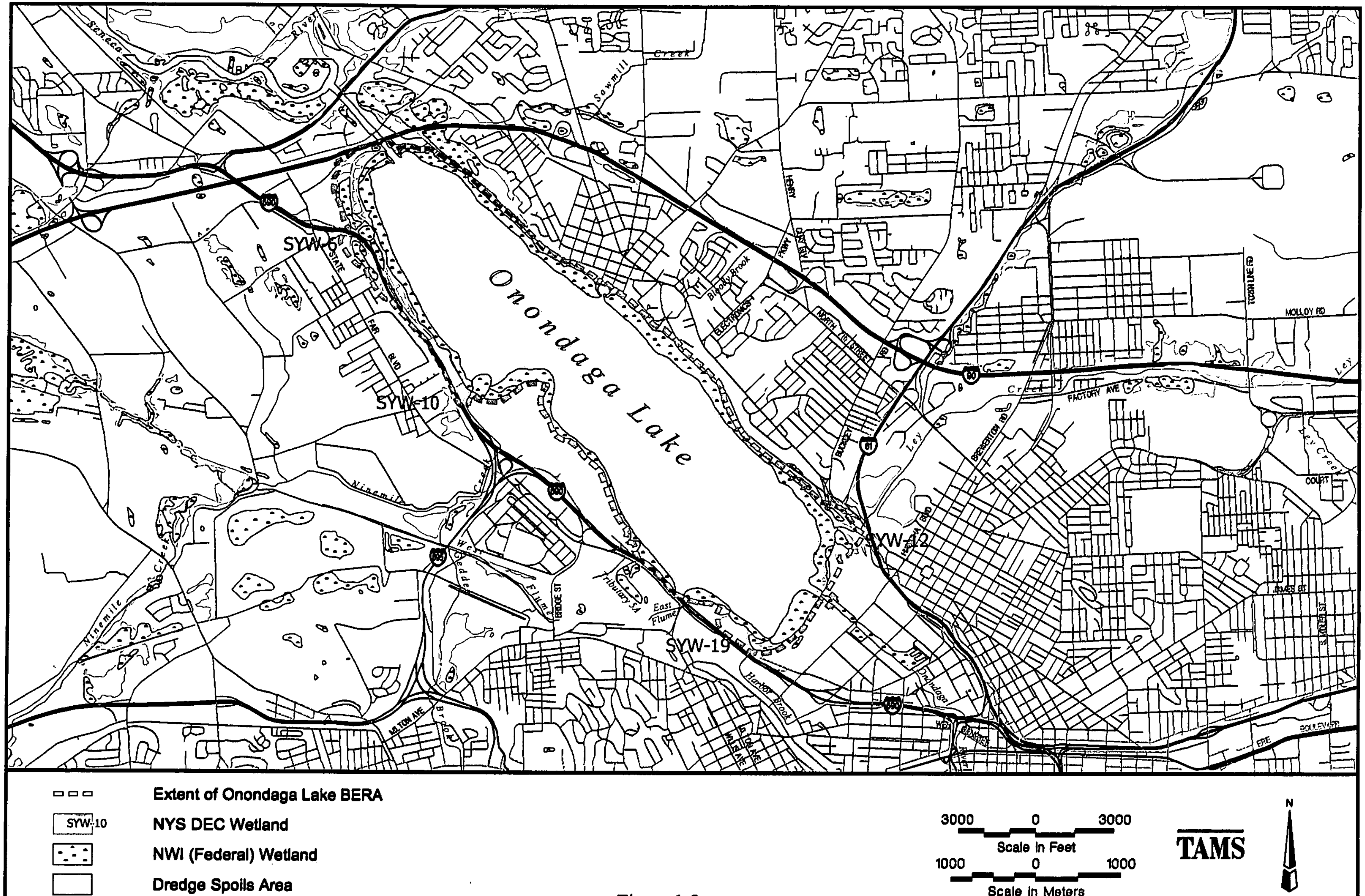
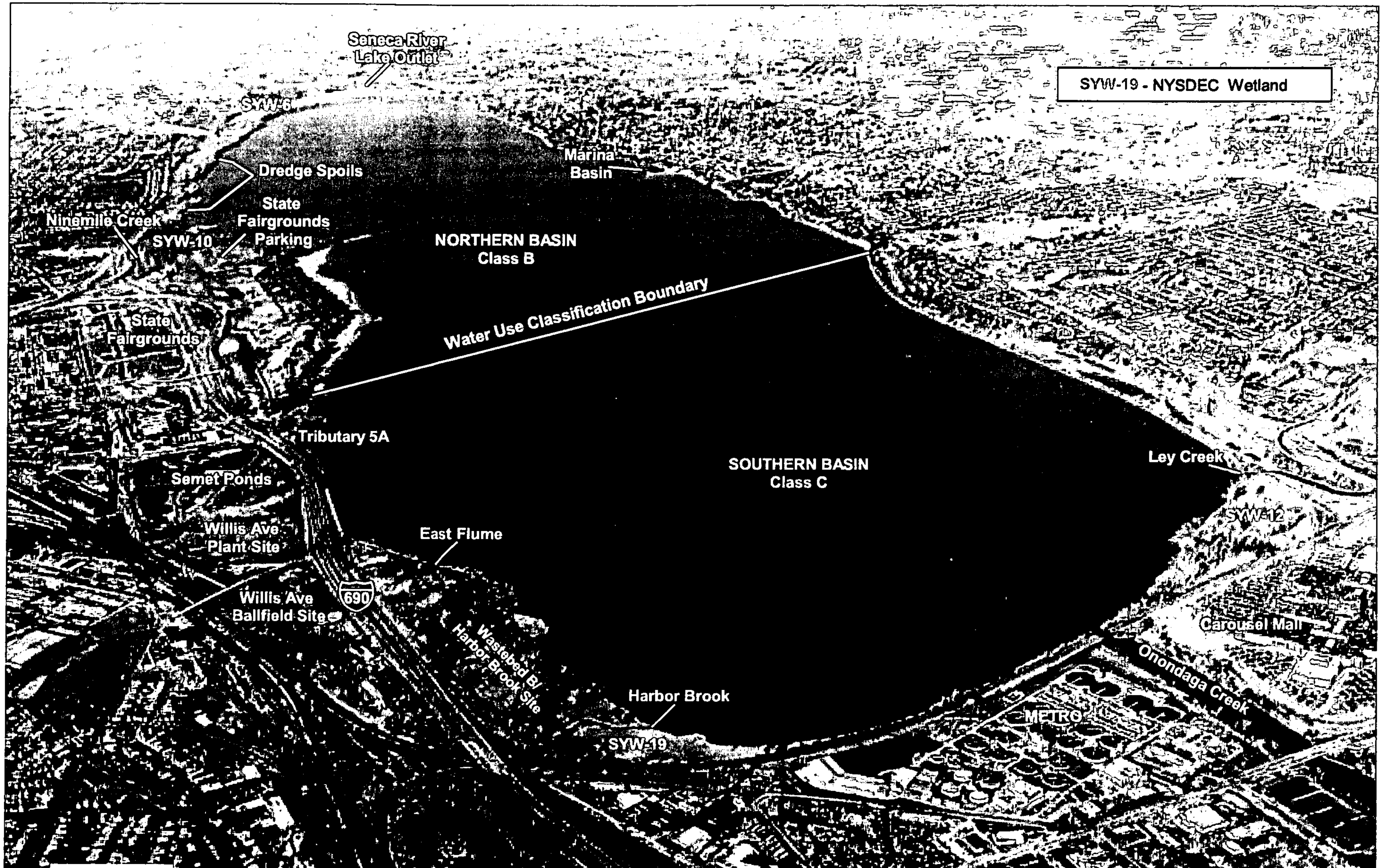


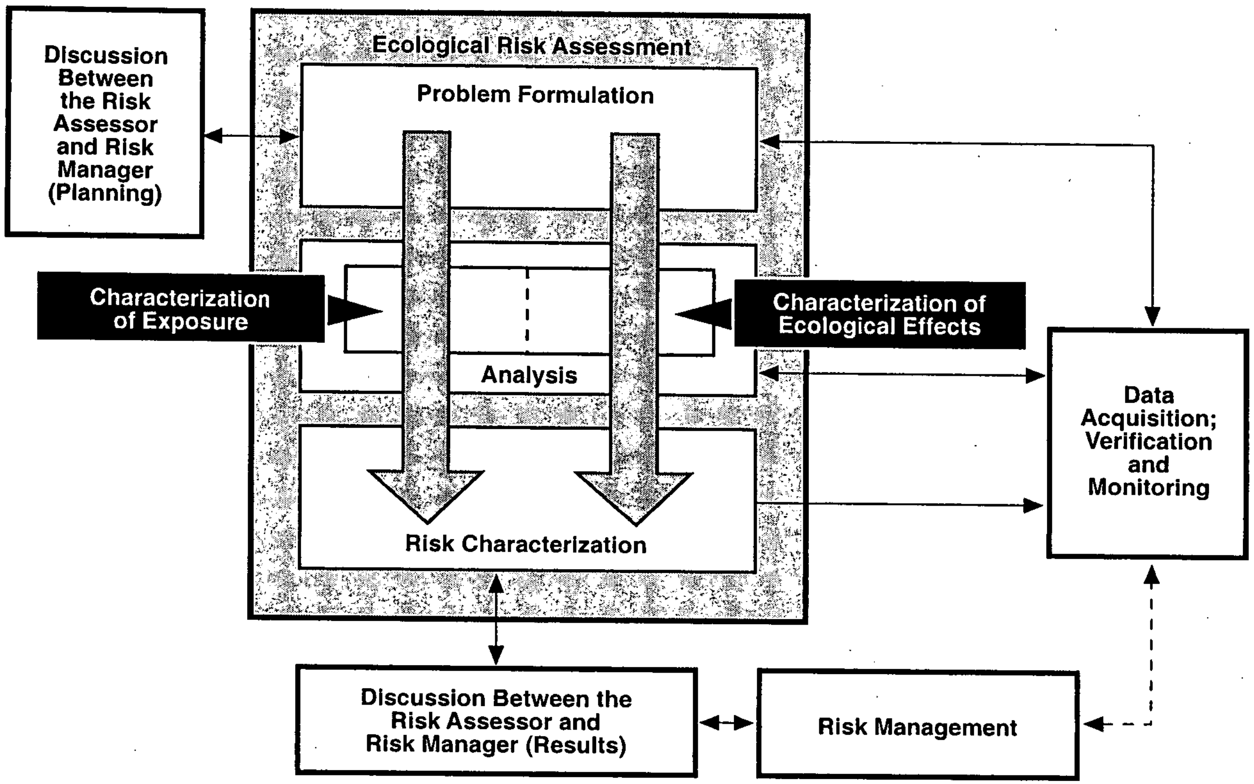
Figure 1-2

Extent of the Onondaga Lake BERA



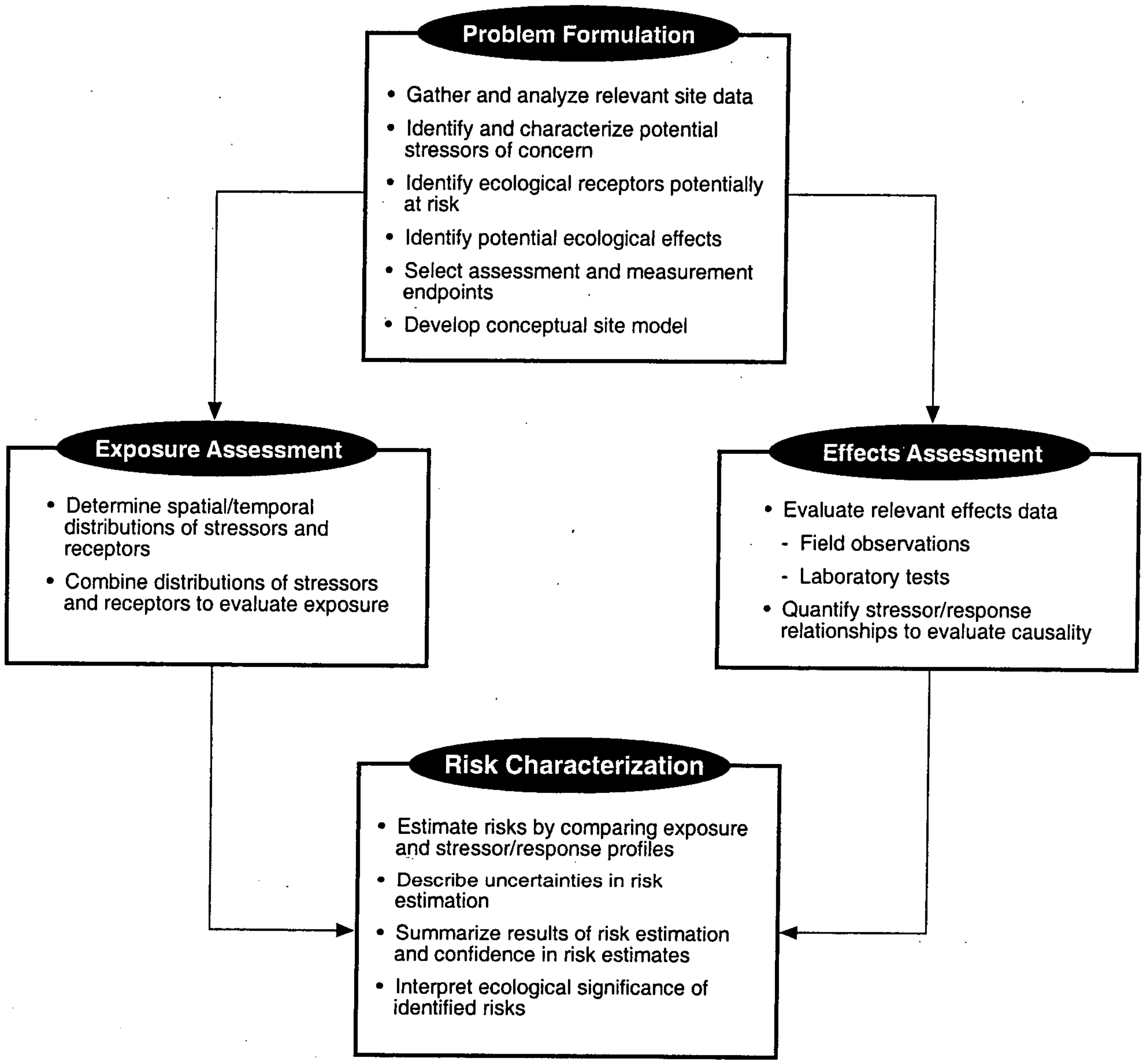
Onondaga Lake BERA

Figure 1-3
Onondaga Lake and Vicinity
Note: figure not to scale



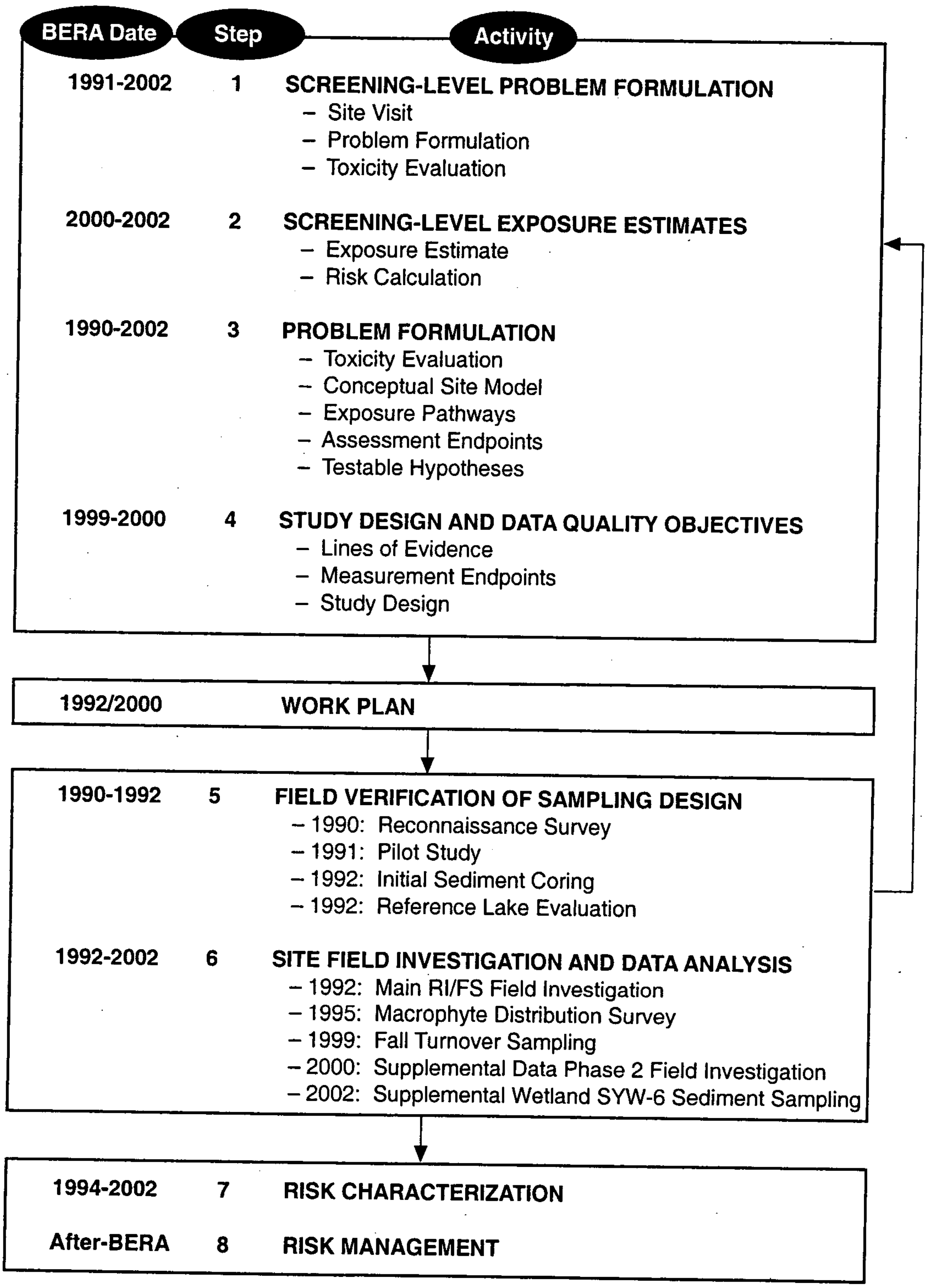
Source: U.S. EPA (1998)
Exponent, 2001b

Figure 1-4. Guidance for conducting ecological risk assessments



Source: Exponent, 2001b

Figure 1-5. Major components of the baseline ecological risk assessment for Onondaga Lake



Source: U.S. EPA (1997a)
Modified from Exponent, 2001b

Figure 1-6. Superfund Guidance for Conducting Ecological Risk Assessments and Relationship to the Onondaga Lake Baseline Ecological Risk Assessment

2. SUMMARY OF HONEYWELL AND OTHER INDUSTRIAL FACILITIES AND ENVIRONMENTAL INVESTIGATIONS

The various Honeywell and other industrial facilities and related areas near Onondaga Lake are briefly described in this chapter, and the major Honeywell environmental investigations that are being conducted are summarized. Additional factors related to the potential transport pathways from the Honeywell upland and tributary sites and non-Honeywell sites to Onondaga Lake are also summarized. Additional information related to these facilities and potential sources of contamination are found in Appendix G of this BERA and in Chapter 4 of the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

2.1 Overview of Honeywell Facilities and Operations

Honeywell's predecessor companies began manufacturing operations in Solvay, New York, in the late 1800s (Figure 2-1). Natural deposits of salt and limestone were the primary reasons for locating the facilities in Solvay. The Solvay Process Company, founded in 1881, used the ammonia soda (Solvay) process to produce soda ash, a product used in a variety of applications such as neutralization, detergent, and industrial chemicals manufacturing and glass manufacturing. Honeywell (through its predecessor corporation, AlliedSignal) subsequently expanded the operation to three locations known as the Main Plant, the Willis Avenue Plant, and the Bridge Street Plant. These three locations are collectively described as the Syracuse Works in this report. The Syracuse Works closed in 1986. Figure 2-2 shows periods of production and production milestones for major product lines at the Syracuse Works.

The Syracuse Works had three major product lines, as follows:

- **Soda Ash** – The soda ash product line primarily produced light and dense soda ash (Na_2CO_3) and a variety of related products, including sodium bicarbonate ($NaHCO_3$, or baking soda), sodium nitrite ($NaNO_2$), ammonium bicarbonate (NH_4HCO_3), ammonium chloride (NH_4Cl), calcium chloride ($CaCl_2$), sodium sesquicarbonate ($Na_2CO_3 \bullet NaHCO_3 \bullet 2H_2O$, or "snowflake"), and caustic soda ($NaOH$).

- **Chlor-alkali** – The chlor-alkali product line primarily produced liquid chlorine, caustic soda ($NaOH$), and caustic potash (KOH). In addition, potassium carbonate (K_2CO_3) and potassium bicarbonate ($KHCO_3$) were produced by carbonating caustic potash. Hydrogen gas was produced as a byproduct of the chlor-alkali process and was used in the manufacture of hydrogen peroxide (H_2O_2) and as a fuel in the power section of the Main Plant.

- **Benzene, Toluene, Xylenes, and Chlorinated Benzenes** – The benzene, toluene, and xylenes product line produced benzene, toluene, and xylenes; heavy hydrocarbons (tars); and naphthalene. The chlorinated benzenes product line produced chlorobenzene, liquid and crystal paradichlorobenzene, liquid and emulsified orthodichlorobenzene, and trichlorobenzenes. Hydrochloric or muriatic acid (HCl) was a marketed byproduct of the chlorinated benzene product line and was also used to lower the pH of feed brine in the chlor-alkali processes.

The Main Plant manufactured soda ash (and related products); benzene, toluene, and xylenes; and naphthalene, whereas the Willis Avenue plant manufactured chlorinated benzenes and chlor-alkali products. The Bridge Street plant produced chlor-alkali products and hydrogen peroxide.

In addition to the three main product lines, Honeywell facilities produced coke and producer gas (a mixture of carbon monoxide, nitrogen, hydrogen, methane, carbon dioxide, and oxygen) for a limited time and generated electricity and steam for use in the manufacturing processes. Several products (i.e., nitric and picric acids; salicylic acid and methylsalicylate; benzyl chloride, benzoic acid, benzaldehyde, and phthalic anhydride; phenol; and hydrogen peroxide) were manufactured for only short periods as either start-up operations that were later relocated or as part of a pilot plant or developmental laboratory activity.

Details about the raw materials, manufacturing processes, and waste materials associated with each of the Honeywell products and activities are presented in the Site History Report (PTI, 1992d). Waste management is also discussed in PTI (1992d) and Blasland & Bouck (1989). Honeywell operated under a variety of National Pollutant Discharge Elimination System (NPDES) and State Pollutant Discharge Elimination System (SPDES) permits.

The wastewater from the Bridge Street plant was discharged to the West Flume, a tributary to Geddes Brook, which in turn is a tributary of Ninemile Creek. Both Geddes Brook and Ninemile Creek are the subject of a separate RI/FS being conducted by Honeywell and NYSDEC. The wastewater from the Main Plant and the Willis Avenue plant was discharged to Onondaga Lake (e.g., via the East Flume; see RI Chapter 4, Section 4.5.1 [TAMS, 2002b]). The East Flume is currently being further evaluated as part of the Wastebed B/Harbor Brook RI/FS being conducted by Honeywell.

The Syracuse Works relied on the use of vast, unlined wastebeds (Solvay Wastebeds) located in the towns of Solvay and Galeville and the city of Syracuse. The locations and designations of the Solvay Wastebeds are shown in Figure 2-3. Initial waste disposal practices consisted of filling wetland areas adjacent to Onondaga Lake. Later, wastebeds designed specifically for Solvay waste disposal were built using containment dikes constructed of materials including native soils, Solvay waste, and cinders or (along the lakeshore) piles and sheeting (Blasland, Bouck & Lee [BBL], 1990).

Several areas near the south end of Onondaga Lake (Wastebeds A through M) contain evidence of Solvay waste disposal (Blasland & Bouck, 1989). Disposal in Wastebeds A through E ceased by 1926, although they received some other materials (e.g., tar residues, sewage sludge) in later years. In particular, Wastebed B between the East Flume and Harbor Brook received significant amounts of Solvay and organic waste either by direct disposal or on-site migration of organic contaminants (TAMS, 2002b). Wastebeds F through M are currently occupied by numerous industrial and commercial structures. Of these, according to Honeywell, only Wastebeds F through H appear to have served as Solvay Wastebeds. Waste material in Wastebeds I through M is probably related to later filling operations associated with road construction (Blasland & Bouck, 1989).

Wastebeds 1 through 8 were used for Solvay waste disposal until 1944. These wastebeds were subsequently transferred to New York State. Disposal in Wastebeds 9 through 11 occurred from 1944 to 1968, and included disposal of Solvay waste, brine purification sediments, and boiler water purification wastes.

Disposal in Wastebeds 12 through 15 began in 1950 and continued until 1986, when the Syracuse facilities were closed. These beds received Solvay waste, brine purification sediments, treated mercury cell wastewater, boiler water purification wastes, and boiler bottom wastes and fly ash. During 1986, the Onondaga County Department of Water Environment Protection (OCDWEP) disposed of liquid sewage sludge (3 to 5 percent solids) and dewatered sludge in Wastebeds 15 and 12, respectively.

The Semet Residue Ponds were disposal lagoons for organic wastes from the Willis Avenue plant. The lagoons were hollowed out of the already-existing Solvay Wastebed A, and filled with approximately 80 million gallons of the tarry residue. The dikes bordering the ponds were reportedly built from fill materials including concrete rubble, old electrolytic cell parts, ashes, cinders, soil, Solvay waste, bricks, stone, etc. (O'Brien & Gere, 1991).

The Wastebed B/Harbor Brook and Willis Avenue Ballfield sites (Figure 2-1) are two additional Honeywell sites that are also currently undergoing investigation. The Wastebed B/Harbor Brook site consists of three areas, including:

- The Lakeshore Area, which was designated as Wastebed B, and received Solvay waste and other industrial wastes from approximately 1908 to 1926, along with additional material in the 1950s.

- The Penn-Can property, which has historically been, and is currently, used for production and storage of asphalt products.

- The CSX Railroad Area, which is located south of the Penn-Can property.

The East Flume and the lower reach of Harbor Brook are also part of the Wastebed B/Harbor Brook site. Previous environmental investigations conducted along Harbor Brook and its vicinity in 1996 and 1997 indicated that mercury, benzene, toluene, ethylbenzene, and xylenes (BTEX), chlorinated benzenes, and polycyclic aromatic hydrocarbon (PAH) compounds are present within the sediments in the lower reach of the brook. Subsequent investigations revealed the presence of non-aqueous phase liquids (NAPLs) on the site and in the lower reach of Harbor Brook. The Willis Avenue Ballfield site, which is the northwest and central portion of Wastebed C, received Solvay waste between approximately 1908 and 1926 (Blasland & Bouck, 1989). The western portion of the Willis Avenue Ballfield site was utilized as a baseball field in the 1960s and 1970s, and possibly as a landfill for Honeywell wastes and debris in the 1940s (O'Brien & Gere, 2000).

The Mathews Avenue Landfill site situated in the Geddes Brook watershed was used by Honeywell as a construction and demolition debris disposal site. A preliminary site assessment (PSA) will be performed at the site.

Based on a review of historic aerial photographs taken from at least 1938 until sometime between 1951 and 1959, NYSDEC has identified that large amounts of Honeywell wastes appear to have been discharged directly to the lake, and, later through the East Flume (TAMS, 2002b). The direct discharge to the lake built up into a delta of waste deposits through which the East Flume now flows. Based on analysis of sediment core samples, these combined wastes included the calcite-contaminated Solvay wastes plus mercury, PAHs, diphenylethanes (including 1-phenyl-1-[2,4-dimethylphenyl]-ethane [PXE] and 1-phenyl-1-[4-methylphenyl]-ethane [PTE]), chlorinated benzenes, and dioxins/furans. At that time, the waste deposits covered approximately 65 acres of the lake bottom (a further discussion of this in-lake waste disposal is provided in Chapter 4 of the RI [TAMS, 2002b]).

Honeywell, in cooperation with Onondaga County, dredged sediments contaminated with mercury from the delta of Ninemile Creek in Onondaga Lake in the late 1960s. The sediments were disposed of in basins constructed in wetlands along the shoreline of the lake just north of the mouth of Ninemile Creek (adjacent to what is now Wetland SYW-10). The location of these basins, referred to as the dredge spoils area, is shown on Figure 2-1 (a further discussion of the dredge spoils area is provided in Chapters 4 and 5 of the RI [TAMS, 2002b]) (A. Labuz, pers. comm., 2000).

2.2 Summary of Non-Honeywell Sources

While Honeywell sites have been important contributors of contaminants to the lake, there are, nevertheless, other industrial facilities in the Onondaga Lake watershed that have, or may have, impacted Onondaga Lake.

There are numerous industrial sites that potentially contributed contamination to Ley Creek, including several landfills, foundries, and other industrial facilities. In addition, the General Motors – former Inland Fisher Guide (GM–IFG) facility is a known contributor of contamination to Ley Creek. The GM–IFG and Ley Creek Deferred Media site, which includes contaminated groundwater associated with the Ley Creek PCBs Dredgings site and surface water and sediments in Ley Creek between Townline Road and Route 11, is being investigated under a separate RI/FS. Ley Creek, below Route 11 near the Town of Salina Landfill, was rerouted in the 1970s. Due to this rerouting, a section of Ley Creek became cut off from the Ley Creek flow (the Old Ley Creek Channel). The sediments and banks of this channel are contaminated with polychlorinated biphenyls (PCBs) and metals (e.g., chromium, cadmium, copper, lead, zinc, and nickel). An RI/FS order is being negotiated with GM for the Old Ley Creek Channel site.

The lakefront area between Ley Creek and Harbor Brook contains several facilities or former facilities that potentially contributed contamination to the lake or Onondaga Creek, including:

- The Oil City area.
- The former Niagara Mohawk Power Corporation manufactured gas plant (MGP) located on Hiawatha Boulevard at the current location of the Metropolitan Syracuse Sewage Treatment Plant (Metro) plant on the south bank of the mouth of Onondaga Creek.
- The former Niagara Mohawk Power Corporation MGP located on Onondaga Creek at Erie Boulevard.
- Metro.
- The American Bag and Metal site on Onondaga Creek.
- Roth Steel near Harbor Brook.

North of Tributary 5A is the Crucible Lake Pump Station disposal site at Crucible Bay. Other industrial facilities, including the Maestri 2 site, may have potentially contributed to contamination in lower Ninemile Creek. Separate RI/FS or other environmental reports have been completed or are currently being prepared for these sites. To the extent that contamination is reaching or has reached Onondaga Lake from these upland sites, the ecological risk associated with that contamination within the boundaries of the Onondaga Lake site is evaluated as part of this BERA.

In the early 1800s, Onondaga Lake was receiving untreated industrial and domestic wastes. Around the turn of the twentieth century, a combined sewer system, a single system that transmits a combination of domestic and industrial flows as well as stormwater originating from various sources, was installed that discharged into tributaries and ultimately the lake.

The first primary sewage treatment facility in the Syracuse area was constructed in 1925 at the southern end of Onondaga Lake. An additional major treatment plant was built in 1940 on Ley Creek. During the 1950s, Onondaga County established a sewer district that encompassed the City of Syracuse and some surrounding suburban areas. A new primary treatment plant, the Onondaga County Metropolitan Syracuse Wastewater Treatment Plant (Metro), was constructed in 1960 with a 50 million gallons per day (mgd) design capacity (Onondaga Lake Management Conference [OLMC], 1993).

The Metro sewage treatment plant, which serves the city of Syracuse and several surrounding towns, is currently permitted (NY-0027081) to discharge an average of 80 mgd through its main outfall to Onondaga Lake. The plant provides tertiary treatment for flows up to 120 mgd. For combined stormwater and industrial/domestic sewage flow up to 220 mgd, the incremental flow above 120 mgd receives primary treatment and seasonal chlorination prior to discharge into the lake through a second outfall.

The sewers contain hydraulic relief structures otherwise known as combined sewer overflows (CSOs), which have historically allowed diluted sewage (due to the mixing of stormwater and sewage) to discharge to several tributaries of Onondaga Lake during high flow events. In 1985, Phase I of a program to abate CSOs was implemented. The second phase of the CSO abatement program began in 1990. Additional abatement activities associated with the CSOs are underway as discussed below.

In January 1998, an Amended Consent Judgment (ACJ) (88-CV-0066) was executed by NYSDEC, the State Attorney General, Atlantic States Legal Foundation, and Onondaga County. The ACJ evolves from a 1989 Judgment on Consent (88-CV-0066) settling litigation between the State of New York and the county relating to state and federal water pollution control regulations.

The ACJ, which is designed to improve the water quality of Onondaga Lake, specifically includes a listing of over 30 projects to be undertaken by Onondaga County over a 15-year period. Although completion of the entire project is not required until 2012, many of these county projects are scheduled for completion by 2009 (OCDWEP, 2002b).

The projects may be grouped into three categories, including:

- Improvement and upgrading of the county's main sewage treatment plant (Metro).
- Eliminating and/or decreasing the effects of the CSOs on the lake and its tributaries.
- Performance of a lake and tributary monitoring program designed to evaluate the effects of the improvement projects on the water quality of the lake and its tributaries.

2.3 Summary of Honeywell's Environmental Investigations

Since closing the Syracuse Works in 1986, Honeywell has been conducting a variety of environmental investigations with the oversight of NYSDEC. The details of each of the major investigations described below are presented in Appendix G, Review of Other Honeywell Sites and Source Areas, including site location, site history, media sampled, maximum detected concentrations of COCs, ecological evaluations, and potential for offsite migration of COCs. Brief summaries of these investigations include:

- **Willis Avenue Chlorobenzene Site** – This investigation addresses the Willis Avenue Plant area and other related areas of study (including the Petroleum Storage Area, the Chlorobenzene Hot Spots Area, and Tributary 5A). The revised RI report was submitted to NYSDEC in October 2002 (O'Brien & Gere,

2002) and is currently under review. The ecological risk assessment for the Willis Avenue site is in progress. A screening-level ecological assessment was submitted to NYSDEC in July 1999 and revised screening tables were submitted in March 2001. A BERA work plan was submitted to NYSDEC in August 1999 and a supplemental biota sampling work plan was submitted in June 2001.

- **Semet Residue Ponds** – This investigation addresses the Semet Residue Ponds. The RI report was submitted to NYSDEC in 1991 (O'Brien & Gere, 1991) and was approved in August 1995. A series of treatability tests were also conducted after approval of the RI and results were reported in O'Brien & Gere (1996, 1997). An FS was submitted to NYSDEC in June 1999 (O'Brien & Gere, 1999b). The proposed plan for the site was issued on January 19, 2002, and the Record of Decision (ROD) was issued on March 28, 2002.

- **LCP Bridge Street Site** – This site is comprised of two separate sites, or operable units (OUs). The OU-1 investigation addresses the former LCP Bridge Street facility and the West Flume, a tributary of Geddes Brook. The draft RI report was submitted to NYSDEC in October 1997 (Gradient and Parsons, 1997), and was subsequently revised and issued as a final report by NYSDEC in August 1998 (NYSDEC/TAMS, 1998a). The draft FS was submitted to NYSDEC in June 1999 (Parsons and Gradient, 1999), a ROD was issued in September 2000 (NYSDEC, 2000b), and a remedial design work plan was approved by NYSDEC on September 18, 2002. An RI/FS is currently underway at the second operable unit (OU-2).

- **Geddes Brook and Ninemile Creek** – This investigation addresses Geddes Brook and the lower reaches of Ninemile Creek, including sediments and floodplain soils. Revised versions of the HHRA, BERA, and RI reports were submitted to NYSDEC in November 2001 (Exponent, 2001d,e,f), rejected by NYSDEC on February 15, 2002, and are currently being rewritten by NYSDEC/TAMS.

- **Wastebeds 1 through 15** – This investigation addresses the wastebeds created by Honeywell along the shorelines of Onondaga Lake and Ninemile Creek. A revised hydrogeologic assessment report for the wastebeds was submitted to NYSDEC in April 1989 (Blasland & Bouck, 1989), and an FS was completed in February 1990 (BBL, 1990). More recently, a supplemental site investigation for Wastebeds 9 through 15 along Ninemile Creek (preliminary site assessment [PSA] complete, Class 2 site) was submitted to NYSDEC in September 1998 (BBL, 1998).

- **Wastebed B/Harbor Brook Site** – A PSA and an RI/FS work plan have been submitted to NYSDEC. Honeywell is currently conducting the RI and associated ecological evaluations (see Section 2.1 for additional information on site areas).

- **Willis Avenue Ballfield Site** – A PSA and an RI/FS work plan have been submitted to NYSDEC. Honeywell is currently conducting the RI and associated ecological evaluations (see Section 2.1 for additional information on site areas).

- **Mathews Avenue Landfill** – The work plan for the PSA for this site was approved by NYSDEC in December 2002.

2.4 Summary of Transport Pathways from Honeywell and Other Sites

From the standpoint of the Onondaga Lake BERA, the most important concern regarding these upland and tributary sites is the potential for offsite migration of COCs and transport to the lake. Based on the information presented in Appendix G and RI Chapter 4 (TAMS, 2002b), the following potential pathways exist for transport of COCs from Honeywell and non-Honeywell sites to Onondaga Lake:

- **Willis Avenue Chlorobenzene Site** – Groundwater and NAPL discharge directly into the lake, as well as surface water transport via the East Flume and Tributary 5A (i.e., following groundwater discharge from the site to those two tributaries). This site is a source of mercury, BTEX, chlorinated benzenes, PAHs, PCBs, and dioxin/furans to the lake.

- **Semet Residue Ponds** – Groundwater and potential NAPL discharge along the lake shoreline, as well as surface water transport via Tributary 5A (i.e., following groundwater discharge from the site to that tributary). This site is a source of mercury, BTEX, naphthalene, and PAHs to the lake.

- **LCP Bridge Street Site** – Surface water transport via Ninemile Creek (i.e., following groundwater discharge from the site to the West Flume and subsequent surface water transport to Geddes Brook and then to Ninemile Creek). This site is a source of mercury, copper, lead, hexachlorobenzene, DDT, benzene, PCBs, and chlorinated solvents to the West Flume and areas downstream.

- **Geddes Brook and Ninemile Creek** – Surface water transport via Ninemile Creek.

- **Wastebeds 1 through 15** – Surface water transport via Ninemile Creek (i.e., following groundwater discharge and surface water transport from the site to the creek) and groundwater discharge from Wastebeds 1 through 8 to Onondaga

Lake. The wastebeds are a source of inorganics, mercury, BTEX, PAHs, and phenols to the lake.

- **Wastebed B/Harbor Brook** – Groundwater and NAPL discharge along the lake shoreline, as well as surface water and NAPL transport via Harbor Brook, and erosion from the shoreline. This site is a source of mercury, BTEX, chlorinated benzenes, PAHs, and phenols to the lake.

- **Honeywell In-Lake Waste Deposit Area** – Wind-induced erosion and resuspension from waste material at the sediment surface, as well as diffusion, bioturbation, and direct contact with biota.

- **Dredge Spoils Area** – This lakeshore area will be further investigated as part of a separate OU. It is currently unknown whether or not this area is a source of contamination to the lake from the dredged material being in direct contact with the local groundwater.

- **Ley Creek** – Ley Creek has received a wide range of contaminants, principally in the form of heavy metals other than mercury and PCBs. These contaminants are also found in the sediments around the mouth of Ley Creek, indicating a contribution from this tributary.

- **Onondaga Creek** – In addition to its sediment load, the creek runs through the city of Syracuse and receives contaminants associated with urban runoff. The American Bag and Metal site is located on both banks of Onondaga Creek and, in the course of a PSA, PCBs were found in the soils on this site, but contaminant migration appears to be minimal. The Niagara Mohawk Erie Boulevard former coal gasification plant, which could also be a source of BTEX or PAHs to the lake, is also located on Onondaga Creek.

- **Oil City** – Industrial compounds utilized and stored in the area included the bulk storage of fuel-related hydrocarbons and the limited location and storage of synthetic organic chemicals and PCBs (Perkins and Romanowicz, 1996). Part of the Oil City area known as the Clark property was remediated under a 1994 ROD that included the installation of a groundwater collection and treatment system for chlorinated and non-chlorinated hydrocarbons, and installation of a containment cell. These contaminants could have historically migrated to the lake or Onondaga Creek. Sediment properties offshore of Oil City contain high levels of PAHs with a pattern that is distinct relative to the naphthalene-dominated pattern at and near the Honeywell sites.

- **Metro and Immediate Area** – Metro, the sewage treatment plant serving the city of Syracuse and certain suburbs, is located on the shore of Onondaga Lake between Onondaga Creek and Harbor Brook. Historically, the Metro facility used wastewater from the Honeywell facilities in order to control phosphorous discharges. It is likely that this use also carried mercury contamination to the Metro facility, making the Metro discharge an inadvertent source of mercury to the lake. Besides the mercury-related discharges, it should be noted that Metro was built on the site of a former MGP (Niagara Mohawk Hiawatha Boulevard site). Residue from such plants typically includes BTEX, PAHs, and cyanides. There is a possibility that these residuals were released into the groundwater at the site, or into the lake, although no evidence of ongoing release has been seen to date. Immediately adjacent to the Metro plant is Roth Steel, which could be a source of metals or PCBs.

- **Sawmill Creek and Bloody Brook** – Sawmill Creek runs through primarily open land and parkland, along with some transportation rights-of-way, and appears not to be a source of COCs to the lake. Bloody Brook runs primarily through a suburban area, some major transportation rights-of-way, and the industrial complex currently owned by Lockheed Martin (Electronics Park). The historic discharges from Electronics Park have contaminated Bloody Brook with cadmium. The sediments and some floodplain areas are to be addressed by removal in a voluntary action conducted by Lockheed Martin and Onondaga County.

As shown by the summaries of potential transport pathways of COCs to Onondaga Lake, the most likely pathways include groundwater discharge along the lake shoreline (i.e., from the Willis Avenue Chlorobenzene site, the Wastebed B/Harbor Brook site, and the Semet Residue Ponds site); resuspension of wastes in the lake; and surface water transport via the East Flume, Tributary 5A, Harbor Brook, and Ninemile Creek.

Groundwater discharges to the lake are currently being evaluated as part of the investigations for the Willis Avenue Chlorobenzene site and the Wastebed B/Harbor Brook site. Other contaminant sources to the lake include Metro, Ley Creek, the Crucible Materials Corporation (via Tributary 5A), and Oil City. A quantitative discussion of fluxes to the lake can be found in Chapter 6 of the RI report (TAMS, 2002b).

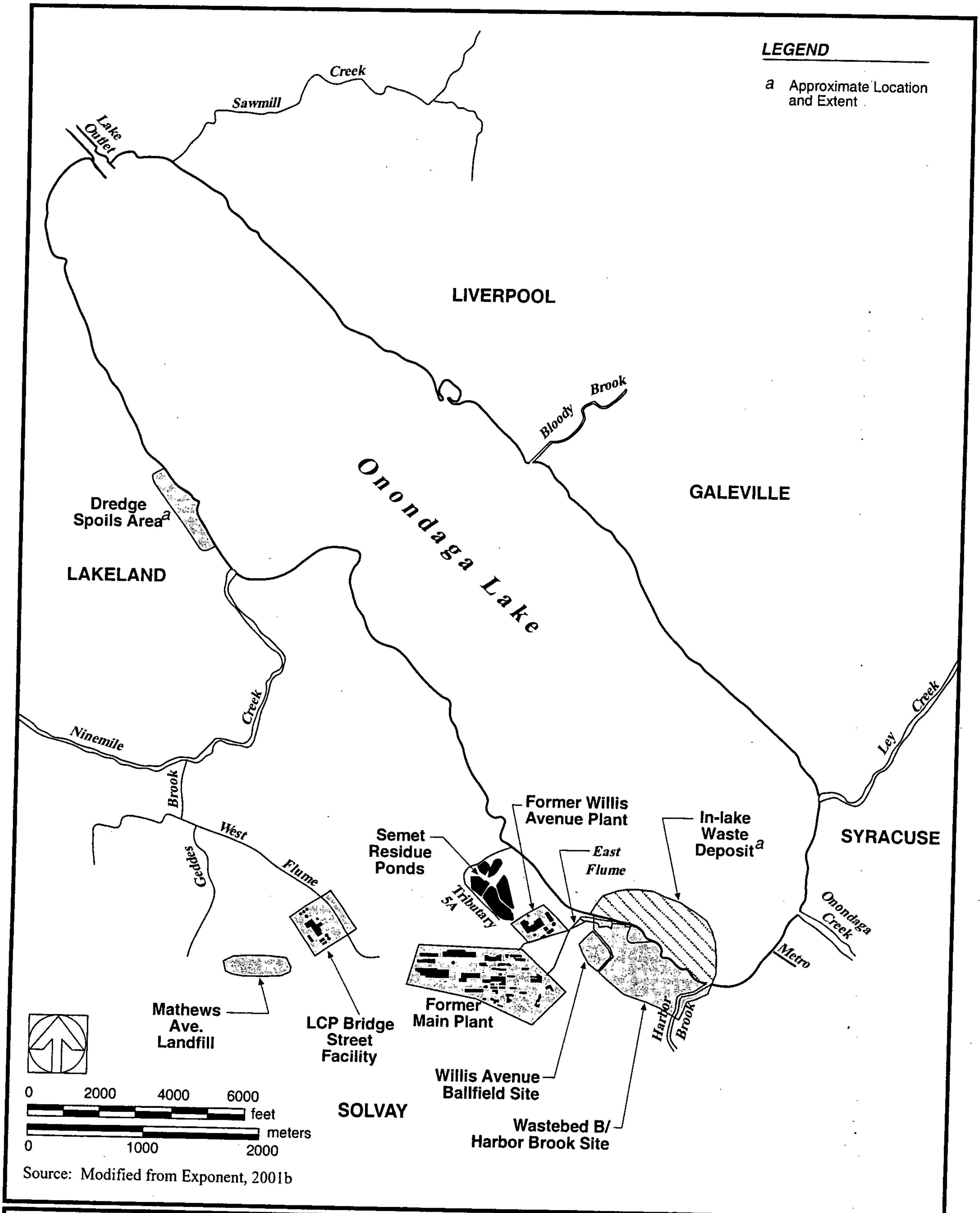
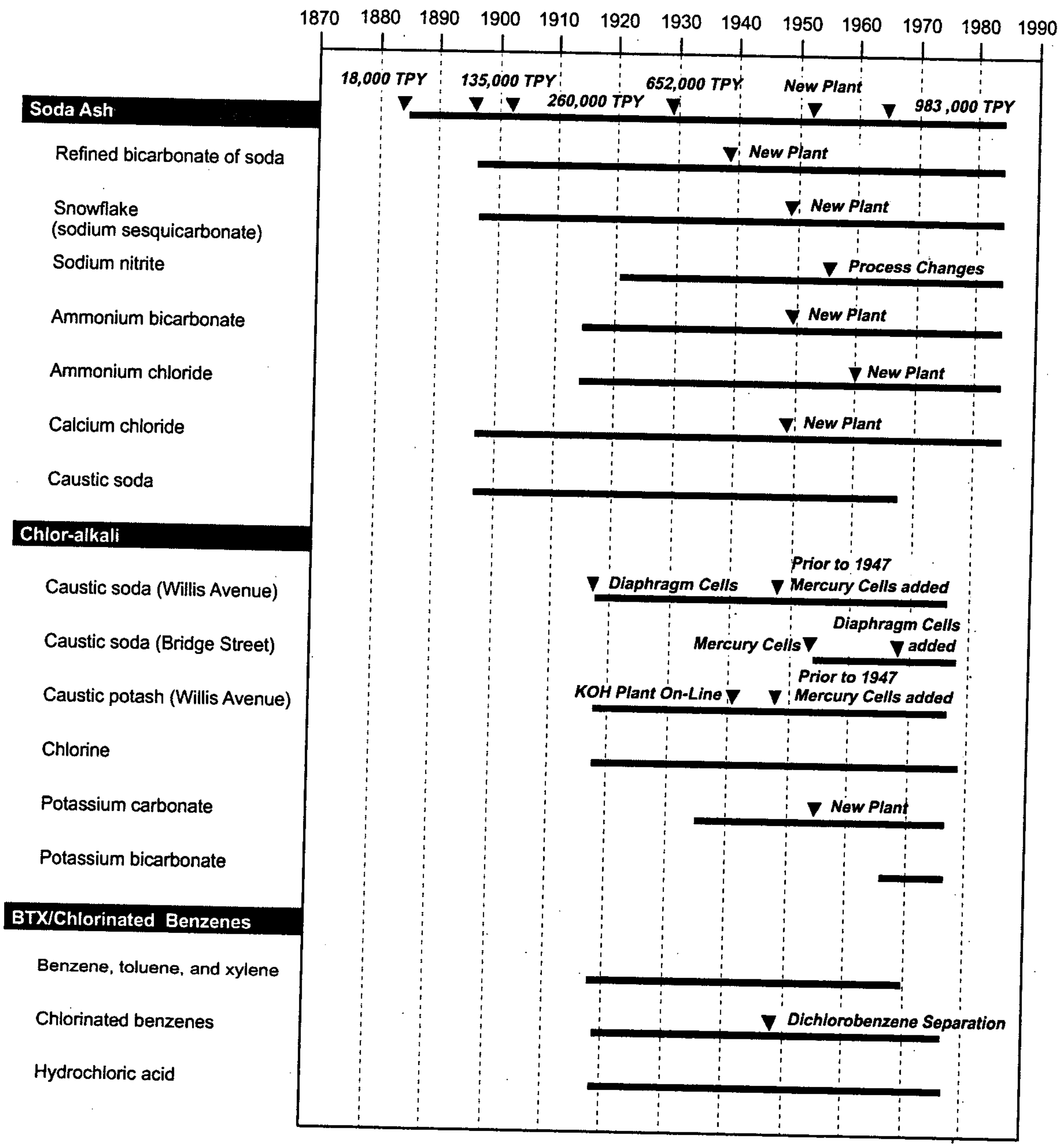


Figure 2-1. Locations of Honeywell Former Facilities and Referenced Disposal Areas near Onondaga Lake



Note: TPY = tons per year

Source: Modified from Exponent, 2001c

Figure 2-2. Periods of Production and Production Milestones for Major Product Lines at the Syracuse Works

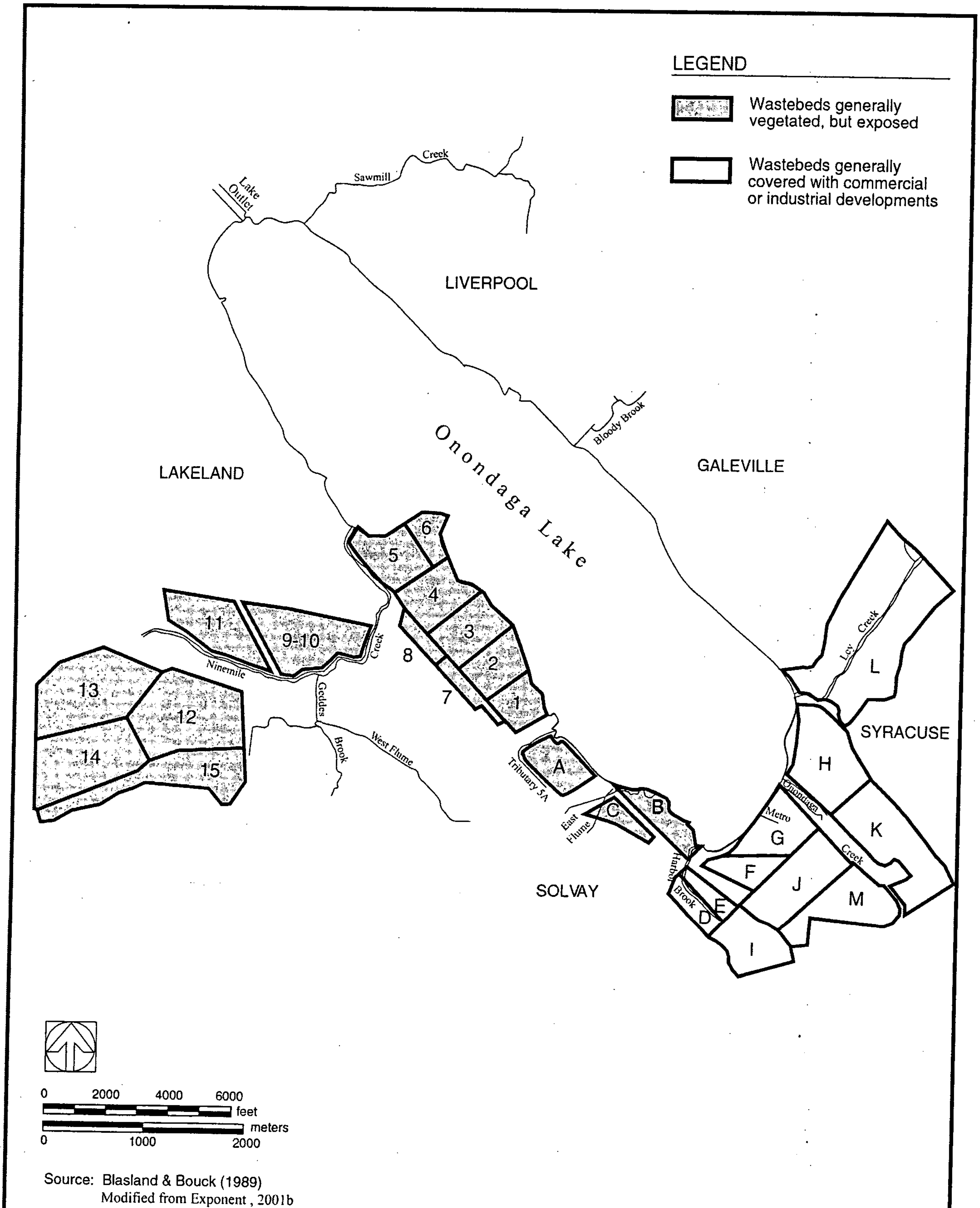


Figure 2-3. Historical Locations of Solvay Wastebeds

3. SITE DESCRIPTION (FWIA STEP I)

This section of the BERA addresses the requirements of Step I of NYSDEC's Fish and Wildlife Impact Analysis (FWIA) for Inactive Hazardous Waste Sites. According to NYSDEC (1994a), the objectives of Step I are to:

- Identify the fish and wildlife resources that presently exist at the site and that existed there before contaminant introduction.
- Provide information necessary for the design of a remedial investigation (RI).

Step I of the FWIA (NYSDEC, 1994a) includes preparation of various site maps, description of fish and wildlife resources, description of fish and wildlife resource values, and identification of applicable fish and wildlife regulatory criteria. The contents of this section are also consistent with the component of Ecological Risk Assessment Guidance for Superfund (ERAGS) Step 1 (see Chapter 4) that addresses the environmental setting (USEPA, 1997a).

3.1 Site Maps

The site maps required for Step I of a FWIA include topographic, wetland, drainage, and covertype (NYSDEC, 1994a). The topographic map for the site is presented in Figure 3-1 and includes the following information:

- Demarcation of the 2-mi (3.2-km) area around the site.
- Topographic features.
- Surface waters (i.e., streams and lakes).
- State and federal wetlands.
- General locations of rare plant species and communities listed in the New York Natural Heritage Program (NYNHP) database.
- Roads and settlements (urban and residential).

As required by FWIA guidance, a drainage map depicting surface flows after hydrological events is presented in Figure 3-2. Wetlands regulated by NYSDEC and those documented by the National Wetlands Inventory (NWI) are presented in Figure 3-3, and are discussed in detail in Section 3.2.4. The covertypes located within 0.5 mi (0.8 km) of Onondaga Lake are presented in Figure 3-4, and are discussed in detail in Section 3.2.4.

3.2 Description of Site Characteristics and Fish and Wildlife Resources

This section describes the physical and biological resources of Onondaga Lake and its surrounding areas.

3.2.1 Lake Morphometry

Onondaga Lake covers an area of approximately 4.6 sq mi (12 sq km), or 3,000 acres, and has a maximum length of 4.7 mi (7.5 km) and width of 1.2 mi (1.9 km) (based on PTI, 1991). The volume of the lake is 139×10^6 m^3. The mean depth of the lake is 12 m, and its maximum depth is 19.9 m. The lake has approximately 11.7 mi (18.8 km) of shoreline (based on PTI, 1991, 1992b). The most recent bathymetric survey of Onondaga Lake was conducted in April 1992 for the RI/FS; the results of this survey are presented in Figure 3-5 (PTI, 1992b). The lake has two basins (northern and southern), which are separated by a slight ridge that is approximately 56 ft (17 m) deep. The maximum depths of the northern and southern basins are 62 and 65 ft (18.8 and 19.9 m), respectively (PTI, 1992b).

As shown in both the bathymetric plot and the hypsographic curve for the lake (Figure 3-5), the nearshore zone of much of the lake at depths less than 4 m is represented by a relatively broad shelf (or bench) bordered by a steep offshore slope at depths of 4 to 8 m.

3.2.2 Climate

The climate in the Onondaga Lake drainage basin can be described as "temperate continental" (Trewartha, 1968) and somewhat humid. The area's geographic proximity to Lake Ontario results in moderated extremes in air temperature, relative to areas at the same latitude that are farther east and are less subject to the "lake effect" (Effler and Harnett, 1996). The mean annual temperature is 48°F (8.8°C), with a mean July temperature of 71°F (22°C) and a mean January temperature of 23°F (−4.9°C) (National Oceanic and Atmospheric Administration [NOAA], 2001). Record temperatures range from 102°F (39°C) in July to −26°F (−32°C) in January, February, and December. Based on data from the period from 1971 to 2000, the average first occurrence of freezing temperatures (daily low of 32°F [0°C]) in the fall is November 15, and the average last occurrence of freezing temperatures in the spring is April 8 (NOAA, 2001).

Moisture enters the area primarily via low-pressure systems that move through the St. Lawrence Valley toward the Atlantic Ocean. Monthly precipitation averages approximately 8.2 cm and is relatively evenly distributed throughout the year, ranging from 6.4 cm in February to 9.4 cm in July (National Climatic Data Center [NCDC], 1995).

Winds in the Syracuse area are predominantly from the west and northwest, as shown in the annual wind rose for the ten-year period prior to 1992 (Figure 3-6). The predominant wind directions remain relatively constant throughout the year, although minor variations occur during different months (Figure 3-7). Most of the strongest winds (20 to 23 m/sec, 44 to 51 mph) occur between November and April (NCDC, 1998).

3.2.3 Geology

Onondaga Lake is located in the southern Ontario Lowlands Province. It is a remnant of ancient Lake Iroquois, a body of water that covered the northern half of Onondaga County 10,000 years ago and included present-day Lake Ontario (Storey, 1977). Onondaga Lake is typical of lakes in the region that were formed by glacial scour approximately 10,000 years ago (NYSDEC, 1989).

Onondaga Lake and most of its drainage basin are located in the Limestone Belt of central New York State (Berg, 1963), a physiographic region that extends from Buffalo eastward to Albany (Figure 3-8). The southern part of the drainage basin is located on the Northern Appalachian Plateau. The surface of some areas in the Limestone Belt consist of deep glacial till derived from limestone and alkaline shales, as well as lacustrine deposits from those materials. Other locations are characterized by outcrops of intact parent strata, particularly Onondaga Limestone. Because most of the water that flows into Onondaga Lake is derived from the Limestone Belt, the soils of the belt have a large influence on the characteristics of the lake water. This influence is particularly apparent for calcium, magnesium, bicarbonate, and alkalinity, the concentrations of which are all higher in lakes influenced by the Limestone Belt than in lakes influenced primarily by the Northern Allegheny Plateau to the south (e.g., the Finger Lakes) or the Ontario-Oneida-Champlain Lake Plain to the north (e.g., Oneida Lake).

Directly underlying Onondaga Lake is Vernon shale, a thick, argillaceous shale. The Syracuse Formation, which is approximately 590 ft (180 m) thick and comprised of shales, dolostones, and salt (Blasland and Bouck, 1989), overlies the Vernon Formation to the south of Onondaga Lake. In this formation, groundwater flows up-dip to the north toward Onondaga Lake and is the source of brines in the area. Brine from the local bedrock also influences water quality in overlying overburden groundwater and in Onondaga Lake tributaries. Kantrowitz (1970) noted that the lower overburden groundwater zones near the lake are influenced by underlying saline groundwater in bedrock.

Pleistocene glaciers extensively eroded the preglacial bedrock and deposited glacial till, which is typically a compact, unsorted, poorly stratified mixture of sands, silt, clay, gravel, and boulders. Till generally overlies the bedrock in this area as a thin veneer about 10 to 16 ft (3 to 5 m) thick. During the time of glacial retreat, large volumes of sediments (glaciolacustrine sediments) accumulated in preglacial lakes. These sediments consist primarily of fine-grained sand and silt, but gravel, coarse-to-medium sand, and clay are present at some locations. More than 245 ft (75 m) of glaciolacustrine sediments were deposited in the southern end of Onondaga Lake (Onondaga County, 1971). In other areas of the Onondaga Lake basin where till and bedrock elevations are higher, glaciolacustrine sediments range from about 15 to 50 ft (5 to 15 m) in thickness.

During the 1992 RI field programs performed by Honeywell as per the Onondaga Lake RI/FS Work Plan (PTI, 1991), sub-bottom profiling revealed about 45 to 60 ft (14 to 18 m) of finer-grained sediment overlying glacial till where acoustic penetration of the sediment was possible in some littoral areas (PTI, 1992b).

3.2.4 Physical Resources

The physical resources of Onondaga Lake described in this section include the major components of both the aquatic and terrestrial environments in and near the lake. The aquatic components include lake water, lake sediment, tributaries, and wetlands. The terrestrial components include soils, the Solvay Wastebeds, and terrestrial covertypes. NYSDEC-designated significant habitats are found in both the aquatic and terrestrial environments of Onondaga Lake.

3.2.4.1 Aquatic Environment

The descriptions of the key components of the aquatic environment in Onondaga Lake are based largely on information presented in the RI/FS Work Plan (PTI, 1991) and on the data collected during Honeywell's 1992 field investigation.

Lake Water

Onondaga Lake is part of the New York State Barge Canal System, and the elevation of the lake is controlled by a dam on the Oswego River at Phoenix, New York, downstream from the lake. Lake elevation can influence numerous characteristics of the nearshore zone because it affects shoreline wetlands, as well as parts of the littoral zone that are subjected to wave and ice disturbance. The mean annual elevation of the lake generally is highest in early spring (due to rainfall and melting snow) and lowest during the summer dry period. From 1971 to 2000, the monthly mean elevation of the lake varied by approximately 1.5 ft (0.5 m) over the annual cycle (Figure 3-9). From 1983 to 1992, the maximum annual variations in lake level ranged from 1.5 ft (0.5 m) (in 1988) to 4.7 ft (1.4 m) (in 1983), with an overall mean of 3.2 ft (0.9 m) for the entire ten-year period (Table 3-1).

The New York State water quality classifications of Onondaga Lake and the lower reaches of its tributaries (6 NYCRR part 701) are presented in Figure 3-10 and include:

- Class B Waters – The lower reaches of Sawmill Creek and Bloody Brook, and most of the northern end of the lake. According to 6 NYCRR Part 701.7, the best uses of Class B waters are primary and secondary contact recreation and fishing. These waters should be suitable for fish propagation and survival.

- Class C Waters – The lower reaches of Harbor Brook, Ninemile Creek, Ley Creek, and Onondaga Creek, the southern end of the lake, and the area of the lake off the mouth of Ninemile Creek. According to 6 NYCRR Part 701.8, the best use of Class C waters is fishing and these waters should be suitable for fish propagation and survival. Class C waters should also be suitable for primary and secondary contact recreation, although other factors may limit the use for these purposes. Tributary 5A is "not classified," but Class C standards apply because it discharges to the southern end of the lake (6 NYCRR Part 895.2).

Like most inland northern lakes, Onondaga Lake is thermally stratified during winter and summer and is isothermal in spring and fall. Stratification governs the distribution of many water-column variables (e.g., water temperature, nutrient concentrations, dissolved oxygen [DO] concentrations) because the thermocline limits vertical mixing between the epilimnion and hypolimnion. During 1992, the thermocline appeared at a depth of approximately 16 ft (5 m) in mid-May and gradually declined to a maximum depth of approximately 43 ft (13 m) by mid-October, when fall turnover occurred (Figure 3-11). In the epilimnion, water temperature reached a maximum value of 72°F (22°C) in mid-June and remained near 68°F (20°C) until the end of September (Figure 3-11). In the hypolimnion, water temperature gradually increased during the period of stratification and reached a maximum of 54°F (12°C) immediately prior to fall turnover.

Prior to 1987, the lake regularly failed to turnover in the spring due to salinity stratification largely caused by manmade influences on the lake (Owens and Effler, 1996). The water inputs from the tributaries affected by the Solvay process tended to plunge into the hypolimnion due to their saline nature and caused a significant saline stratification. The failure of the lake to turnover caused a depletion of the DO in the hypolimnion and prevented the normal heating of these waters (Effler et al., 1996; Owens and Effler, 1996). After the chlor-alkali plant closed in 1986 turnover resumed, although saline inputs from the wastebeds continue to affect stratification. Dissolved oxygen in the hypolimnion is also generally depleted in the late summer or early fall due to manmade eutrophication (Effler et al., 1996).

Lake Sediments

The grain-size distribution and total organic carbon (TOC) content of sediments can be used to infer depositional patterns throughout Onondaga Lake. As shown in Figure 3-12, grain-size distribution and TOC content were closely associated in Onondaga Lake in 1992. The highest percentages of fine-grained sediment (>90 percent) and TOC (>3.0 percent) were found in the deeper parts of both the northern and southern basins. By contrast, the coarsest sediments (<10 percent fine-grained fraction) and lowest TOC values (<1.0 percent) were found (PTI, 1993e) throughout most of the nearshore zone along the entire eastern shoreline and the western shoreline north of Ninemile Creek. The sedimentary patterns in Onondaga Lake in 1992 are similar to the patterns found by others in the lake (Johnson, 1989; Auer et al., 1996b).

Historically, a flocculent layer (estimated to be approximately 17 percent solids by weight) was believed to be present over much of the surface of the sediments (Effler, 1975). Verification of the presence, depth, and extent of the flocculent layer was impeded by the difficulty of recovering core samples from this layer. However, Effler (1975) estimated that the flocculent layer at that time was approximately 50 to 90 cm thick. A flocculent layer was not observed during the 1992 and 2000 field investigations.

Much of the nearshore area of Onondaga Lake is covered with oncolites resulting from the calcium-contaminated discharge of ionic waste into the lake (Dean and Eggleston, 1984), as discussed in Chapter 4, Section 4.1.1.3. Oncolites are irregularly rounded, calcareous nodules that range in size from 0.5 to 30 cm and are not attached to substrates (Pentecost, 1989).

Using cesium-137 as a chemical marker for strata corresponding to the years 1963 to 1964,[1] Saroff (1990) estimated that sedimentation rates for the northern and southern basins of the lake were approximately 0.8 and 0.9 cm/year, respectively, from 1963 to 1990. During the RI/FS, mercury, calcium, cesium-137, lead-210, and pollen were used as markers to estimate sedimentation rates based on core samples taken near the center of each basin. The results of these analyses indicated that the average sedimentation rate from 1972 to 1992 was approximately 0.9 cm/year, with a decrease in sedimentation rate after plant closure in1986.

Tributaries

Onondaga Lake receives surface runoff from a drainage basin estimated to cover approximately 248 sq mi (642 sq km) (Figure 3-13) (Effler and Whitehead, 1996). Surface water flows primarily from the south and southeast into the lake via six tributaries: Ninemile Creek, Onondaga Creek, Ley Creek, Harbor Brook, Bloody Brook, and Sawmill Creek. Water is also discharged to the lake by Metro and through intermittent bidirectional flow from the Seneca River at the outlet of the lake (Effler et al., 1986). In addition, a small amount of water is added to the lake through two industrial conveyances: the East Flume and Tributary 5A.

Together, Ninemile Creek and Onondaga Creek accounted for approximately 62 percent of the total inflow during the period 1971 to 1989 (Figure 3-14) (Effler and Whitehead, 1996). During the same period, the Metro discharge accounted for 19 percent of the total inflow, Ley Creek accounted for 7.7 percent of the inflow, and Harbor Brook accounted for 2.2 percent of the inflow. Contributions by all other tributaries were minor.

The highest inflow of water to Onondaga Lake occur in March and April and the lowest inflow occurs in August (USGS, 1990). Water exits the lake via the outlet at the northwest end and flows into the Seneca River (Figure 3-13). The Seneca River merges with the Oneida River to form the Oswego River, which discharges to Lake Ontario.

Groundwater within the Onondaga Lake drainage basin generally flows from the tributary valleys to the lake, following the topography. The groundwater-flow paths and exchanges with surface water depend on local geologic conditions within each tributary valley. Groundwater within the bedrock discharges to the lake through a number of natural brine seeps along the southwest and southern portions of the lake and along the various tributary valleys leading to the lake (Blasland and Bouck, 1987).

Seneca River

The Seneca River is a large river that drains approximately 3,500 sq mi (9,000 sq km) of central New York to the Oswego River, and subsequently to Lake Ontario. Much of the river is part of the Barge

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[1] Large quantities of cesium-137 were released into the atmosphere in 1963, the year before the Nuclear Test Ban Treaty went into effect (1964).

Canal, and flows and water levels are regulated accordingly. The Seneca River receives all of the outflow from Onondaga Lake via the lake outlet.

Near the confluence with the lake outlet, the water column of the Seneca River is sometimes affected by salinity stratification, which results largely from ionic discharges from the lake (Canale et al., 1996; Owens and Effler, 1996). Relatively dense lake water often exits along the bottom of the lake outlet, as river water flows into the lake in the upper levels of the channel. This occurrence is promoted by the elevated salinity of the lake water and the absence of a natural hydraulic gradient between the lake and river (a condition that resulted from the historical channelization of the lake outlet to support navigation).

Stratification in the Seneca River has been identified as a reason for reduced concentrations of dissolved oxygen in the river near the confluence with the lake outlet (Canale et al., 1995, 1996). In July 1991, the stratification extended more than 5 mi (8 km) downstream from the lake outlet (Canale et al., 1995, 1996). The stratification is generally limited to periods of low flow in the river, because turbulence during high flow is sufficient to break up the stratification.

Wetlands

There is little information regarding the original condition of the wetlands surrounding Onondaga Lake. Onondaga County is noted for conditions that lead to the formation of marl fens (i.e., peatlands with the water table usually at or just above the surface) (Olivero, 2001). The marl found in the soil and sediments surrounding the lakes suggest that some of the original wetlands surrounding the lake were marl fens. The remnant inland salt ponds and marshes, and historical accounts of salt springs on the lakeshore, suggest that inland salt marshes were also present in the area surrounding the lake (Effler and Harnett, 1996). The total extent of wetlands was likely affected when the level of Onondaga Lake was lowered by about 2 ft (0.6 m) in 1822 (Effler and Harnett, 1996). In addition, development and waste disposal by Honeywell (formerly AlliedSignal) along the southern and southwestern shoreline (e.g., in the vicinity of Wastebeds 1 through 8) has buried much of the original wetland habitats.

NYSDEC and federal wetlands (based on NWI maps) currently located within 2 mi (3.2 km) of Onondaga Lake are shown in Figure 3-3. Characteristics of these NYSDEC wetlands, such as class, area, and predominant vegetation, are presented in Table 3-2. NYSDEC classifies and regulates wetlands in New York State pursuant to 6 NYCRR Parts 663 and 664. Regulated wetlands must be at least 12.4 acres (5.02 hectares) in area and must be dominated by hydrophytic vegetation. Smaller wetlands having "unusual local importance as determined by the Commissioner" may also be regulated by the state.

Twenty-two state-regulated wetlands exist either wholly or partially within 2 mi (3.2 km) of Onondaga Lake (see Figure 3-3) (NYSDEC, 1986). Four of these wetlands occur along or near the lake's shoreline near the mouths of Harbor Brook, Ley Creek, and Ninemile Creek, as well as along the northwest shoreline of the lake. These four wetlands, SYW-6, 10, 12, and 19, are directly connected to Onondaga Lake and are therefore believed to be representative of the impact of the lake's contamination on wetlands in its vicinity.

- **Wetland SYW – 6,** located at the northwest border of Onondaga Lake, is a 100-acre (40.6-hectare), Class I wetland. The wetland is divided by a series of elevated paths. The paths are used primarily by pedestrians, but are large enough to support vehicles. The paths create cells in the wetland that are not obviously connected by surface flows, though some cells are hydrologically connected via culverts. A few cells in this wetland are directly connected to the lake through culverts under the paths. The cells in the wetland vary in vegetation type but are dominated by floodplain forest or emergent swamps.

- **Wetland SYW – 10,** located along Ninemile Creek, is a 27.2-acre (11-hectare), Class I wetland. This wetland is divided by Interstate 690 (I-690). On the lake side of I-690 the wetland is dominated by emergent vegetation and floodplain forest. This portion of the wetland is also being investigated as part of the Geddes Brook/Ninemile Creek site. The wetland section on the western side of I-690 was originally a salt marsh; however, the saline inputs appear to be gone and the wetland is now dominated by typical emergent vegetation. A portion of the wetland on the western side of I-690 (known as the "Maestri 2" site) has been filled with waste from the Crucible Materials Corporation and is currently being independently investigated as part of a separate RI/FS by the potentially responsible parties (PRPs) for the Maestri 2 site.

- **Wetland SYW – 12,** located between the mouth of Onondaga Creek and the mouth of Ley Creek, is a 40.7-acre (16.5-hectare), Class I wetland. The northeast edge of this wetland is separated by railroad tracks. As the wetland approaches the lake it is dominated by emergent vegetation. Along the shore of the lake the wetland is a combination of floodplain forest and emergent marsh.

- **Wetland SYW – 19,** located at the mouth of Harbor Brook, is a 19.8-acre (8-hectare), Class II wetland. This wetland is currently dominated by reedgrass (*Phragmites australis*), a species commonly found in disturbed or contaminated areas. This wetland area is located on Honeywell property and is also being investigated as part of the Wastebed B/Harbor Brook site. A discussion of the Wastebed B/Harbor Brook site as a source of contamination to Onondaga lake is discussed in Appendix G of this BERA and in the Onondaga Lake RI (TAMS, 2002b).

The US Fish and Wildlife Service (USFWS) maintains maps of wetlands and deepwater systems through the NWI program. NWI-identified wetlands may be any size and fall within the jurisdiction of the US Army Corps of Engineers (USACE). A total of 205 individual NWI wetlands and deepwater systems occur within 2 mi (3.2 km) of Onondaga Lake (see Figure 3-3) (USFWS, 1999), including 3 limnetic lacustrine systems, 15 littoral lacustrine wetlands, 2 low-perennial riverine systems, and 185 palustrine wetlands.

Table 3-3 presents the physical and biological attributes of each of the above-listed wetlands or systems, according to the NWI classification scheme (USFWS, 1999).

NYSDEC Significant Habitats

According to the database maintained by the NYNHP, sensitive aquatic habitats located near Onondaga Lake are inland salt ponds and marshes (NYNHP, 2001, 2002; see Appendix C). As described below, inland salt ponds are found along the southeastern shoreline of the lake (see Figure 3-4). Inland salt marshes are found adjacent to the inland salt ponds, as well as along Ninemile Creek, west of I-690 (see Figure 3-4).

3.2.4.2 Terrestrial Environment

The following descriptions of the key components of the terrestrial environment near Onondaga Lake are based largely on the information presented in the RI/FS Work Plan (PTI, 1991).

Soils

The soils of the Onondaga Lake watershed consist primarily of glacial till mixed with glacial outwash, alluvial deposits, and unconsolidated sediments. The soils tend to be medium-textured, well drained, and high in lime (NYSDEC, 1989; Soil Conservation Service [SCS], 1977). The drainage basin of the lake is in the northern portion of a region of drumlins and is characterized by narrow, steep-sided valleys. During rainstorms, large amounts of soil erode into valley streams (Lincoln, 1982; Murphy, 1978; NYSDEC, 1989).

Most of the soils along the western, southern, and eastern sides of the lake have been so substantially altered by humans that the original soils are unrecognizable or absent. These soils are classified as "made land" and "urban land" (NYSDEC, 1989). The urban land includes developed areas covered by concrete and buildings, such as parking lots, business parks, and shopping malls. In addition, land has been created along the southern half of the lake shoreline by filling areas with sand, silt, brick, ashes, cinders, Solvay waste, and other wastes.

Solvay Wastebeds

The soda-ash wastes generated as part of the soda-ash manufacturing process at the Honeywell facilities were deposited in a series of wastebeds along the southern and western shorelines of Onondaga Lake and along Ninemile Creek (PTI, 1991). The wastebeds located on the western shoreline are currently exposed (see Chapter 2, Figure 2-3), whereas some of the remaining wastebeds have been covered. The Solvay wastes also extend into the lake in some areas. Vegetation has begun to colonize some of the wastebeds; however, in many areas the vegetation is sparse and composed of few species. In areas where the slope of the wastebed is steep, vegetation is unable to grow and exposed cliffs are visible. Along the southwest shoreline of the lake, cliffs have formed due to the erosion of the waste into the lake.

Lakeshore

In general, the eastern shore of Onondaga Lake is urban and residential, and the northern shore is dominated by parkland, wooded areas, and wetlands. The northwest upland is mainly residential, with interspersed urban structures and several undeveloped areas. Much of the western lakeshore is covered by wastebeds, and, to a lesser extent, dredge spoils from the lake, many of which have been abandoned and recolonized by vegetation. Urban centers and industrial zones dominate the landscape surrounding the south end of Onondaga Lake from approximately the fairgrounds to Ley Creek. More detailed descriptions of the covertypes found along various parts of the lakeshore are presented below.

The terrestrial covertypes found within 0.5 miles (0.8 km) of Onondaga Lake were presented previously in Figure 3-4. The covertypes were mapped using a combination of aerial photographs and the results of ground-level surveys. Approximately 42 percent of the areal extent of covertypes identified in Figure 3-4 is residential, 33 percent is urban/industrial, and 25 percent is characterized as open, forested, or palustrine. Detailed descriptions of each kind of covertype are in Appendix A. Characteristic flora of each covertype community are listed in Table A-1 of Appendix A.

East

Urban development associated with the city of Syracuse and the towns of Liverpool and Galeville characterizes the eastern shore of Onondaga Lake. Onondaga Lake Parkway (Highway 370) and railroad tracks pass very close to the southern portion of the eastern shore. The middle section of the eastern shoreline includes a marina, public landing, and Onondaga Lake Park. The parkland follows the shoreline north to the lake outlet, and similar habitat (mowed lawn with trees) extends north along the east side of the outlet to the Seneca River. Several segments of the former Oswego Canal still exist within the park.

Shallow emergent marsh and marsh dominated by reedgrass and purple loosestrife (*Lythrum salicaria*) border Ley Creek, which flows into the southeast corner of the lake. Successional open habitat (old fields and shrubland) and unmowed roadside areas characterize the area lying northwest of Ley Creek, to the east of the Oswego Boulevard Parkway. Several inland salt ponds surrounded by salt marsh and successional shrubland occur along the southern section of the eastern shore.

Northwest

The Sawmill Creek area north of the lake is generally low-lying and dominated by reedgrass/purple loosestrife marshland and floodplain forest. These communities extend northwest toward the Seneca River and are bisected below John Glenn Boulevard by a stand of successional northern hardwoods. The open residential lands lying to the east of the wooded areas are characterized by mowed lawn with trees.

The predominant vegetative community on Klein Island (at the mouth of the lake outlet) is successional northern hardwood forest. Two stands of floodplain forest visible in aerial photographs suggest that lower areas are subject to periodic flooding. However, this land is private and could not be accessed for verification.

The area extending west from the northern section of the lake outlet toward the Seneca River and following the river west for at least 0.5 mi (0.8 km) is primarily wetland and includes floodplain forest, shallow and deep emergent marshes, and reedgrass/purple loosestrife marsh. A mixture of habitats, including open residential land, stands of successional northern hardwoods, patches of floodplain forest, and a pine plantation, lie between John Glenn Boulevard and I-90 on the west side of the lake outlet. From here, an industrial zone extends southwest on either side of I-90.

Onondaga Lake Park extends down the northwest shore of the lake from the lake outlet to Ninemile Creek and includes both wetlands and adjacent forested upland areas. The shoreline here is characterized by wetland communities, including reedgrass/purple loosestrife marsh, floodplain forest, deep emergent marsh, and, in particular, silver maple/ash swamp. Several patches of open water occur throughout this zone. The upland communities to the southwest of the park are mainly residential, with a few forested areas as well as smaller patches of trees and successional open habitat.

Lakeview Point Area (West)

The mouth of Ninemile Creek forms the northern border of an area known as Lakeview Point. A reedgrass/purple loosestrife community and floodplain forest follows the creek bed upstream, southwest of the lake. Lakeview Point is comprised mainly of calcareous waste derived from soda-ash production, although several of the wastebeds here were also used as landfills for steel-mill waste and sewage sludge disposal (PTI, 1991). Some vegetation has colonized these wastebeds despite the poor nutrient content of the calcareous substrate (Richards, 1982). One section of Lakeview Point was seeded with grass in the recent past.

A large section of Lakeview Point serves as a parking area for the state fairgrounds and is classified as unmowed roadside habitat. South of Lakeview Point are the fairgrounds themselves, surrounded by unmaintained lawn, pavement, mowed roadside, and urban structures. Several successional old fields and a stand of successional northern hardwoods lie beyond the railroad tracks south of the fairgrounds. East of this area, the terrain is covered with a mixture of urban structures, urban vacant lots, successional shrubland, and pavement. Another inland salt marsh, considerably larger than those remaining on the east side of the lake, lies just west of the north end of Lakeview Point, to the west of I-690.

South

Reedgrass/purple loosestrife marsh dominates the lakeshore south of Lakeview Point. The mouths of Tributary 5A, the East Flume, and Harbor Brook lie within this marsh, as does a strip of successional shrubland. Trees and shrubs follow Harbor Brook upstream, south of the marsh. The upland portion of the southwestern shoreline includes wastebeds, urban vacant lots, successional shrubland, mowed roadside, and several interconnecting railroad tracks. Tributary 5A is surrounded by shrubland.

The old Barge Canal terminal is at the mouth of Onondaga Creek. The lakeshore to the west of the terminal is riprap, while a mixture of reedgrass/purple loosestrife marsh and floodplain forest populate the shoreline between the terminal and Ley Creek.

An entirely industrial area lies to the west of the canal terminal. The ground cover in that area includes junkyards, maintenance spoils depositories, the Metro sewage-treatment plant, urban structures, mowed and unmowed roadsides, pavement, and interstate highways.

Urban structures and an old field form most of the west bank of the canal terminal. A large regional shopping mall is located immediately to the north of the canal terminal and west of the Oswego Boulevard Expressway. A large paved area is located southeast of the mall, also alongside the canal terminal. North of the expressway and southeast of Ley Creek is another urban area comprising mowed and unmowed roadside, old fields, mowed lawn, and urban structures.

NYSDEC Significant Habitats

According to the database maintained by the NYNHP, there are no significant or sensitive terrestrial habitats near Onondaga Lake (see Appendix C).

3.2.5 Biological Resources

The key biological resources described in this section include the major communities of aquatic, semiaquatic, and terrestrial organisms, including rare, threatened, and endangered species, found in and around Onondaga Lake, as follows:

- Major aquatic communities – macrophytes, phytoplankton, zooplankton, benthic macroinvertebrates, and fish.
- Major semiaquatic organisms – amphibians and reptiles.
- Major terrestrial organisms – plants, birds, and mammals.

These groupings are used for general descriptions of biological resources. However, there are exceptions to these broad characterizations. For example, some snakes may spend their entire life cycle in upland areas and some birds and mammals (e.g., loon [*Gavia immer*] and river otter [*Lutra canadensis*]) may spend most of their time in aquatic habitats.

3.2.5.1 Aquatic Species

Macrophytes

Little information is available on the historical occurrence of macrophytes in Onondaga Lake. There are accounts of macrophyte beds at the northern end of the lake, near the mouths of tributaries, and immediately south of the discharge point of Tributary 5A (Murphy, 1978). It has also been reported that *Potamogeton pectinatus* (Stone et al., 1948), *P. crispus* (Saroff, 1990), and *Ceratophyllum* sp. (Saroff, 1990) have been observed in the lake at various times. Dean and Eggleston (1984) suggested that extensive beds of charophytes (either *Nitella* sp. or *Chara* sp.) were present in the lake because the stems of those plants formed the nuclei of the majority of the oncolites that are found throughout parts of the nearshore zone of the lake. The disappearance of charophytes may be attributed to the enriched calcium discharge associated with the Solvay process (Dean and Eggleston, 1984).

The most recent studies of macrophytes in Onondaga Lake have been conducted between 1991 and 1995 (Madsen et al., 1993, 1996; Auer et al., 1996a; PTI, 1993c; Arrigo, 1995). The six species identified in the lake during those studies are *Ceratophyllum demersum*, *Elodea canadensis*, *Heteranthera dubia*, *Myriophyllum spicatum*, *Potamogeton crispus*, and *P. pectinatus*. The distribution of macrophytes throughout the nearshore zone of Onondaga Lake was mapped in 1992 during the RI using aerial photography and ground-level verification of the photographic results (Figure 3-15). The survey was repeated in 1995 using identical methods to evaluate potential changes in macrophyte distributions during the three-year period between surveys.

In 1992, five macrophyte species were identified in the lake (all species noted above except *Elodea canadensis*). Although macrophyte beds were found throughout the littoral zone of the lake, relatively large areas of the littoral zone were characterized by sparse beds. Major beds were largely confined to the southeastern corner of the lake (between Harbor Brook and Onondaga Creek) and along the eastern and northern shorelines between Bloody Brook and the lake outlet. In 1995, an additional species (*E. canadensis*) was identified in the lake and the distribution of macrophyte beds had expanded throughout the nearshore zone of the lake. Major new beds were found (Arrigo, 1995) near the mouths of Ninemile and Ley Creeks, and a series of new beds was found off the western shoreline north of Tributary 5A.

Phytoplankton

Phytoplankton communities have been routinely monitored at two stations in Onondaga Lake since 1970 by the Onondaga County Department of Water Environment Protection (OCDWEP). One station is located near the center of the northern basin of the lake, and the other station is located near the center of the southern basin. In addition to the monitoring studies, Sze and Kingsbury (1972) and Sze (1975, 1980) conducted evaluations of phytoplankton communities in the lake. Murphy (1978) has reviewed much of the historical information on phytoplankton communities of Onondaga Lake. More recent information is presented in Auer et al. (1996a).

In 1992, 36 phytoplankton taxa were collected (PTI, 1993c; Stearns & Wheler, 1994) in Onondaga Lake (Table 3-4). According to Stearns & Wheler (1994), blooms of phytoplankton in 1992 continued to be a symptom of the eutrophic condition of the lake. The major algal groups in 1992 were flagellated green algae, non-flagellated green algae, diatoms, cryptomonads, and cyanobacteria (i.e., blue-green algae). Since 1986, non-flagellated green algae and diatoms have declined in abundance, whereas cryptomonads and flagellated green algae have continued to be abundant. Between 1990 and 1992, phytoplankton abundances remained relatively constant. However, there has been an overall decline in the abundance of eukaryotic algae, coupled with an increase in abundance of blue-green algae. From April to June of 1992, phytoflagellates were dominant until a clearing event with low algal abundances began on June 3. Blue-green algae were abundant from mid-July until early September.

Zooplankton

In addition to phytoplankton, zooplankton communities in Onondaga Lake have been routinely monitored since 1970 by OCDWEP. Murphy (1978) briefly reviewed a subset of this historical information. The zooplankton communities of the lake have also been evaluated by Meyer and Effler (1980), Garofalo and Effler (1987), Auer et al. (1990, 1996a), and Siegfried et al. (1996).

Between 1986 and 1989, 25 zooplankton taxa were collected in Onondaga Lake (Table 3-5) (Auer et al., 1996a). Zooplankton communities were dominated by cladocerans, copepods, and rotifers. From 1990 to 1992, zooplankton abundances remained relatively constant. In 1992, three peaks of copepod and cladoceran abundances were found: late May to June, late July to August, and late September to November (PTI, 1993c; Stearns & Wheler, 1994). Historically, Onondaga Lake has had a very low number of zooplankton species (Auer et al., 1996a). The total number found during the 1986 to 1989 study demonstrated a large increase in the number of species found in the lake. However, even this increased number of species is small when compared to other lakes in the region. Contamination of lake water by stressors and chemicals is the likely cause of the lack of species richness in the lake (Auer et al., 1996a).

Benthic Macroinvertebrates

Historically, the characteristics of benthic macroinvertebrate communities in Onondaga Lake have not been intensively studied. For the historical studies that have been conducted, sampling was performed at only a small number of stations in the lake. Results of the historical evaluations have been described by Stone et al. (1948), Noble and Forney (1971), and Auer et al. (1996a).

In 1989, the populations of benthic macroinvertebrates in the lake were dominated by pollution-tolerant species of oligochaetes and chironomids. While the species richness was low, the density of local population was high. Several pollution-intolerant species that should be expected in similar unpolluted lake environments were absent, including crayfish, caddisflies, and mayflies (Auer et al., 1996a).

In 1992, benthic macroinvertebrate communities were sampled at 68 stations throughout Onondaga Lake as part of the RI (PTI, 1993c). More than 70 taxa were identified in the samples (Table 3-6). Communities at most stations were dominated numerically by oligochaetes and chironomids. The lake's benthic communities were sampled at 15 stations in 2000, with a similar number of taxa identified (i.e., more than 70). Communities continued to be dominated numerically by oligochaetes and chironomids. The benthic macroinvertebrate communities sampled in 1992 and 2000 are described in greater detail in Chapter 9, where they are used as indicators of potential sediment toxicity.

Fish

Historically, Onondaga Lake supported a cold-water fishery. Common species found in the lake included Atlantic salmon (*Salmo salar*), cisco (*Coregonus artedii*), American eel (*Anguilla rostrata*), and burbot (*Lota lota*) (Auer et al., 1996a). The first scientific survey of the lake in 1927 indicted that the cold-water fishery was disturbed due to the impacts of soda-ash production (Table 3-7). By 1969, the fishery of Onondaga Lake was described as a warm-water fishery with none of the cold-water species observed. Historical information on the fish communities of Onondaga Lake has been collected by Greeley (1927), Noble and Forney (1971), and Chiotti (1981). The most current information on fish communities in the lake has been summarized by Gandino (1996) and Auer et al. (1996a).

According to Auer et al. (1996a) and Tango and Ringler (1996), Onondaga Lake supports a warm-water fish community that is dominated by the pollution-tolerant gizzard shad (*Dorosoma cepedianum*), freshwater drum (*Aplodinotus grunniens*), carp (*Cyprinus carpio*), and white perch (*Morone americana*). Sunfish are abundant in the littoral zone. The lake supports several important sportfish, including channel catfish (*Ictalurus punctatus*), largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieui*), and walleye (*Stizostedion vitreum*).

Between 1927 and 1994, 57 species of fish were collected in Onondaga Lake (Table 3-7). The abundance of these species varies widely. Eleven species, including banded killifish (*Fundulus diaphanus*), bluegill (*Lepomis macrochirus*), brook silverside (*Labidesthes sicculus*), carp, emerald shiner (*Notropis atherinoides*), gizzard shad, golden shiner (*Notemigonus crysoleucas*), largemouth bass, pumpkinseed (*Lepomis gibbosus*), white perch, and yellow perch (*Perca flavescens*), appear to reproduce with moderate or high success around the lake (Table 3-8). All the remaining species of fish have limited or no success reproducing in the lake (Auer et al., 1996a).

The Oswego River basin supports 100 species of fish, including burbot, green sunfish (*Lepomis cyanellus*), trout-perch (*Percopsis omiscomaycus*), and brook trout (*Salvelinus fontinalis*). Individuals of these pollution-intolerant species (Tango and Ringler, 1996) are seldom encountered in the lake and there is no evidence of these species reproducing within the lake.

A number of species collected in Onondaga Lake migrate in from other areas. For example, the one lake trout caught in Onondaga Lake was a tagged fish that originated from a stocking in the Finger Lakes (Tango and Ringler, 1996). Species that are not known to reproduce in the lake are dependant on other areas to maintain the population within the lake. Ringler et al. (1995) conducted a tagging study of lake fish in 1990 and 1991, and found that a number of fish migrated out of the lake and entered the Seneca River system. Tagged fish were found as far upstream as Baldwinsville on the Seneca River (6.2 mi [10 km] away) and as far downstream as Fulton on the Oswego River (15.5 mi [25 km] away). The authors also used radio telemetry to follow fish movements during fall turnover in 1991. Several fish were found to leave the lake and enter the Seneca River during the turnover period (i.e., when DO concentrations become reduced in lake water). At least one individual later moved back into the lake when DO concentrations increased. Ringler et al. (1995) concluded that the Seneca River is a corridor for fish movement into and out of Onondaga Lake. The authors also noted that these movements indicate that some fish with elevated chemical concentrations in tissue likely leave the lake and enter the Seneca River system.

Rare, Threatened, and Endangered Aquatic Species

According to the databases maintained by the NYNHP and USFWS, there are no federally or state-listed rare, threatened, or endangered aquatic species in Onondaga Lake (see Appendix C).

3.2.5.2 Semiaquatic Species

The species associated with the original wetlands surrounding the lake were not recorded. Cicero Swamp, a large wetland in Onondaga County, is home to a large number of unique wetland species such as the eastern Massasauga rattlesnake (*Sistrurus catenatus*) and the spotted turtle (*Clemmys guttata*). Because the conditions of most of the wetlands surrounding the lake are not similar to Cicero Swamp, the species occurring in the swamp have not been considered as species to be expected near the lake. However, it is possible that conditions for these species were historically present around the lake.

The amphibian and reptile species expected to occur in the habitats surrounding Onondaga Lake are listed in Table 3-9. Surveys of the amphibians inhabiting Onondaga Lake and its surrounding wetland and terrestrial habitats were conducted by researchers from the State University of New York (SUNY) Cortland from 1994 to 1997 (Ducey and Newman, 1995; Ducey, 1997; Ducey et al., 1998). In addition, species distributions and qualitative evaluations were conducted from March 1995 to May 1997 (Ducey, 1997). Although the surveys were directed toward the assessment of amphibian populations, reptiles were also identified and recorded when encountered.

The amphibian and reptile species found near Onondaga Lake between 1994 and 1997 are listed in Table 3-10 and include the following taxa:

- **Amphibians** – Seven species, comprised of five species of anurans (i.e., frogs and toads) and two species of salamanders.

- **Reptiles** – Six species, comprised of three species of aquatic snakes and three species of turtles.

Fewer species were found around Onondaga Lake than expected. In general, the numbers of amphibian and reptile species found near the lake were less than the numbers typically found in similar areas of central

New York State (Ducey and Newman, 1995; Ducey, 1997; Ducey et al., 1998). Amphibian reproduction appears to be limited to wetlands that are not directly connected to Onondaga Lake water (Ducey, 1997), indicating intolerance of lake water. Additional discussion on potential toxicity is included in Chapter 9.

Rare, Threatened, and Endangered Semiaquatic Species

According to the databases maintained by the NYNHP and USFWS, there are no federally or state-listed rare, threatened, or endangered semiaquatic species in Onondaga Lake (see Appendix C).

3.2.5.3 Terrestrial Species

Birds

Onondaga Lake provides a variety of habitats for bird species. Table 3-11 lists species of birds found around Onondaga Lake during the Breeding Bird Atlas survey conducted from 1980 to1985 (Andrle and Carroll, 1988). More recent data suggest that additional species have started to use the lake. The recent Breeding Bird Atlas survey (beginning in 2000 and scheduled to extend until 2004) has recorded other species, such as the turkey vulture (*Cathartes atratus*) and Canada goose (*Branta canadensis*) (Table 3-11). A list of 22 additional bird species observed near Onondaga Lake in the summer of 1993, but not identified in the Andrle and Carroll 1988 survey (and, therefore, not presented in Table 3-11), is presented in Table 3-12 (Tango, 1993).

A list of 13 species of waterfowl that overwintered near Onondaga Lake between 1990 and 1999 is presented in Table 3-13. The New York Audubon Society conducts annual winter surveys (Christmas counts by county) and waterfowl surveys (by water body).

Mammals

A list of 45 mammalian species that potentially occur near Onondaga Lake is presented in Table 3-14. Some of the more common species include opossums, shrews, rodents, muskrats, raccoons, skunks, and deer. New York State species of special concern and endangered species are also identified in this table.

Rare, Threatened, and Endangered Terrestrial Species

Ten state-listed and one federal rare, threatened, or endangered species have been observed near Onondaga Lake. They include four plant species and seven species of birds (see Appendix C).

The three listed plant species within 2 mi (3.2 km) of Onondaga Lake are Sartwell's sedge (*Carex sartewelli*), little-leaf tick-trefoil (*Desmodium ciliare*), and red pigweed (*Chenopodium rubrum*). All three plant species are known only from historical records. They have not been sighted in the Onondaga Lake area recently, but may be rediscovered. Hart's tongue fern (*Asplenium scolopendrium var* *americanum*), a federally listed threatened species, may also be present in the area of Onondaga Lake. The general locations of listed plants near Onondaga Lake are shown on Figure 3-1.

The six state-listed bird species of special concern observed near Onondaga Lake (Tables 3-11 and 3-12) are the common loon (*Gavia immer*), osprey (*Pandion haliaetus*), sharp-shinned hawk (*Accipiter striatus*), common nighthawk (*Chordeiles minor*), red-headed woodpecker (*Melanerpes erythrocephalus*), and horned lark (*Eremophila alpestris*). The common tern (*Sterna hirundo*) is classified as a New York State-threatened species.

3.2.6 Observations of Stress

As specified for Step I of an FWIA (NYSDEC, 1994a), any atypical biotic conditions observed at a site should be identified. Although there have been numerous observations of potential stress in Onondaga Lake in past studies, some of the more recent observations of potential stress on lake biota include:

- Reduced species richness and standing crop of macrophytes in the nearshore zone (Auer et al.,1996a).
- Blooms of nuisance forms of cyanobacteria (i.e., blue-green algae) in the water column during summer (Auer et al., 1996a).
- Increased oncolite density (Dean and Eggleston,1984).
- Chloride loadings to Onondaga Lake from Wastebeds 1 to 8 through seeps on the east side of the beds, deep groundwater discharges, and direct erosion of Solvay waste by wave action along the lakefront (Blasland & Bouck, 1989).
- Reduced species richness of zooplankton communities (Auer et al.,1996a).
- Dominance of benthic macroinvertebrate communities by pollution-tolerant taxa (Auer et al., 1996a).
- Apparent lack of reproduction in the lake by numerous fish species (Auer et al., 1996a).
- Change in fishery assemblage from cold-water fishery to a warm-water fishery dominated by pollution-tolerant species (Tango and Ringler, 1996).
- Mercury contamination of fish (NYSDEC, 1987).
- Disappearance of fish from the lake during fall turnover (Auer et al., 1996a).

- Reduced species richness of amphibians and reptiles (Ducey et al., 1998).

- Lack of amphibian reproduction in wetlands directly connected to lake water (Ducey, 1997).

- Lack of spring turnover in the lake prior to 1987 (Owens and Effler, 1996).

3.3 Description of Fish and Wildlife Resource Values

As specified for Step I of an FWIA (NYSDEC, 1994a), a qualitative assessment of the value of fish and wildlife resources at a site should be made with respect to both associated fauna and humans. The current conditions and potential future value of contaminant-free resources are discussed below.

3.3.1 Value to Associated Fauna

3.3.1.1 Wetlands

Current Conditions

There are approximately 320 acres (130 hectares) of state-regulated wetlands and numerous smaller wetlands directly connected to Onondaga Lake or within its floodplains (i.e., Wetlands SYW-1, SYW-6, SYW-10, SYW-12, and SYW-19). The value of the wetlands currently connected to the lake has been reduced by the contamination in the lake. The disturbance due to contamination is evident in the limited breeding and decrease in amphibian populations. Lack of waterfowl species identified in the Breeding Bird Atlas for 1980 to 1985 (Andrle and Carroll, 1988) suggested that wetlands around the lake were not being used extensively by waterfowl. The effects of wetland contamination on fish and mammal populations has not been evaluated.

Potential Future Value

Recent improvements in water quality within the lake may be allowing the return of wildlife populations that utilize the wetlands. Also, improvements to the water quality in the lake may improve the wetland habitat quality itself and facilitate the return of additional species. Continued improvement of sewage treatment, closure of the Honeywell plant on the western shore of the lake in 1986, eliminating sources of pollution, and potential lake and wetland remediation projects will permit the wetlands to be more suitable for a variety of species.

The wetlands surrounding the lake can function as breeding habitat for waterfowl and other birds. Recent sightings of increased numbers of waterfowl and shoreline-related birds suggest that the populations may be beginning to recover. The unique saline character of the lake allows for the formation of inland salt marshes, a globally rare habitat. The wetlands surrounding the lake could provide breeding habitat for many species of amphibians and turtles. Fish may use the wetlands directly connected to the lake as breeding habitat; these same wetlands can function as fish nurseries.

3.3.1.2 Aquatic Habitats

Current Conditions

Lake water and sediment contamination has reduced the value of the lake to resident aquatic life, and is having a significant effect on a number of aquatic and semi-aquatic populations. Many fish are unable to reproduce in the lake, due to either breeding habitat degradation or the lack of DO. The fish are contaminated with chemicals, which are passed on to their predators. Lakewide populations of macrophytes are small, due to an inability to anchor in many parts of the lake. The low quality of lake sediments contaminated by ionic wastes and the presence of oncolites make macrophyte establishment difficult, resulting in wave action having an amplified effect on aquatic macrophytes in the lake.

Despite widespread contamination, wildlife populations continue to use the lake. Onondaga Lake is within the Atlantic flyway and functions as a stopover point for mergansers, loons, and other waterfowl migrating to the Adirondack Mountains. The fish populations within the lake provide the basis for the diets of many species. Osprey, gulls, herons, terns, and cormorants regularly feed on fish in Onondaga Lake. Presently many species of fish are dependent on other areas, such as the Seneca River, for breeding habitat to support their populations.

Potential Future Value

In the past, Onondaga Lake has supported salmonid species such as the Atlantic salmon. The upper reaches and tributaries of Ley Creek, Harbor Brook, and Ninemile Creek are Class C waters with C(T) standards (for trout waters), and could potentially provide breeding habitat for salmonids. Research suggests that continued recovery and restoration of aquatic plants could lead to greater reproductive success for the species of fish within the lake.

3.3.1.3 Terrestrial Habitats

Current Conditions

Many locations along the lakeshore area have been heavily urbanized and contaminated, which has reduced the value of the lake to resident terrestrial species. The urbanization and the levels of contamination are having a significant effect on a number of terrestrial populations which inhabit the lakeshore.

Potential Future Value

The shores of Onondaga Lake provide habitat for several mammal species. Recovering populations of otter appear to be moving toward the lake (NYSDEC, 2002a; Stiles, 2001). Woodchuck (*Marmota monax*), muskrat (*Ondatra zibethicus*), and squirrels (e.g., *Sciurus carolinensis*) are regularly observed on the shores of Onondaga Lake. These and other small-mammal species support predators such as mink (*Mustela vison*), fox (*Vulpes fulva* and *Urocyon cinereoargenteus*), and coyote (*Canis latrans*). The less-disturbed shoreline of the northwest section of the lake can provide habitat for more reclusive or larger species, such as beaver (*Castor canadensis*) and deer (*Odocoileus virginianus*).

Typically, large bodies of water in urban areas provide important habitat to migrating bird species, which use the lakeshore as a resting area during migration. Reductions in contamination are also likely to improve the distribution and abundance of terrestrial invertebrates, reptiles, and amphibians, which are major food sources for many terrestrial vertebrates, such as birds and small mammals. Populations of reptiles and amphibians may become reestablished in the Onondaga Lake area.

3.3.2 Value to Humans

The fish and wildlife resources of Onondaga Lake and its surrounding areas are used by humans for a variety of purposes, including:

- **Boating** – The marina located on the eastern shoreline of the lake, and the lake's connection to the Seneca River, facilitate use of the lake by boaters. In addition, Syracuse University maintains a boathouse on the lake outlet and uses the lake for competitive rowing events. The lake is connected to the Barge Canal system and an effort has been made to encourage boaters using the canal to stop at the lake.

- **Fishing** – Onondaga Lake contains numerous fish species, such as walleye, largemouth bass, and smallmouth bass, that are sought after by recreational anglers. Fishing was banned in Onondaga Lake in 1970 due to elevated mercury levels in fish flesh. It was reopened as a catch-and-release fishery in 1986, with an advisory to eat no fish (Sloan et al., 1987). In 1999, restrictions were lifted to current levels, which state that individuals should consume no more than one meal per month of Onondaga Lake fish, with the exception of walleye, which should not be eaten at all due to elevated levels of mercury, and that women of childbearing age and children under the age of 15 should eat none (NYSDEC, 2002b). Recent fishing derbies have been held to increase public interest in the fishery. In the past other species of fish were present in the fishery such as cisco, salmon, and trout.

- **Hunting and Trapping** – Where permission has been granted by the appropriate landowner and when laws permit, the shores of Onondaga Lake provide hunting and trapping opportunities. Waterfowl and deer populations are abundant enough to support hunting. In addition, mink, fox, and other mammals can be trapped.

- **Recreation** – More than 75 percent of the shoreline of Onondaga Lake is owned by Onondaga County and is classified as parkland, which in 1990 was used by more than one million people. Common activities include picnicking, walking, jogging, roller blading, and bicycling. A project is now underway to construct a recreational-use path around the entire lakeshore.

- **Swimming** – In the past, the lake has been used for swimming. The northern two-thirds of the lake is classified by New York State for direct recreational contact (i.e., Class B Waters). Swimming is limited due mainly to the lack of permitted beaches.

- **Inner Harbor** – New development has occurred near the southern shore of Onondaga Lake, along Onondaga Creek and the Barge Canal, as part of the Syracuse Inner Harbor Project. Approximately 42 acres of land, owned by the New York State Canal Corporation, are being developed for recreational and commercial uses by the Lakefront Development Corporation (LDC).

- **Commerce** – Onondaga Lake has long served as a backdrop for a number of industrial and commercial sites. Historically, the shores of Onondaga Lake were extensively developed by industries. The central location in the state, the salt deposits, and the presence of water supported an extensive salt recovery industry. Other industries also developed around the lake, some of which are still in operation today. Industrial sites have been converted to develop commercial properties in the vicinity of the lake, including Carousel Mall, the Regional Farm Market, and the New York State Fairgrounds parking area.

- **Tourism** – The city of Syracuse, Onondaga County, and New York State are attempting to increase the tourism industry in Syracuse. A future expansion of Carousel Mall, the development of the Inner Harbor, and the lakeside bike path are all part of this effort. The lake is central to these efforts as a scenic and recreational area.

- **Stormwater Retention** – The lake and its surrounding wetlands and tributaries are used extensively by Onondaga County for stormwater retention and discharge.

There are several plans to further enhance the use of Onondaga Lake and its surrounding areas by humans. For example, the Onondaga Lake Management Conference (OLMC) has prepared a management plan for the lake (OLMC, 1993), and the State of New York, Onondaga County, and the city of Syracuse have prepared a development plan for the lake (Reimann-Buechner Partnership, 1991).

3.4 Identification of Applicable Fish and Wildlife Criteria

Step I of an FWIA (NYSDEC, 1994a) requires the identification of both contaminant-specific and site-specific criteria applicable to the remediation of fish and wildlife resources. Section 121(d) of

Comprehensive Environmental Response Compensation and Liability Act of 1980 (CERCLA) requires that remedial actions comply with state and federal applicable or relevant and appropriate requirements (ARARs). Applicable requirements are defined as any standard, requirement, criterion, or limitation promulgated under federal environmental law or any promulgated standard, requirement, criterion, or limitation under a state environmental or facility siting law that is more stringent than the associated federal standard, requirement, criterion, or limitation.

Relevant and appropriate requirements are those cleanup standards, control standards, and other substantive environmental protection requirements, criteria, or limitations promulgated under federal or state law that, while not "applicable" to a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstance at a National Priorities List (NPL) site, address problems or situations sufficiently similar (relevant) to those encountered, and are well-suited (appropriate) to circumstances at the particular site. Requirements must be both relevant and appropriate to be ARARs.

Potentially applicable laws and regulations, fish and wildlife criteria, and benchmark values are summarized in this section for use in the screening evaluation for the Onondaga Lake BERA. Selection of the screening values was based on their applicability to either freshwater or terrestrial environments. The potential chemical-specific, location-specific, and action-specific ARARs for evaluation in the Onondaga Lake FS in each of the three categories, along with other to-be-considered (TBC) requirements, are summarized in Chapter 9, Section 9.2 of the RI (TAMS, 2002b; Tables 9-1 to 9-6).

3.4.1 New York State Laws and Regulations

- **New York Environmental Conservation Law (ECL), Article 15, Title 3 and Article 17, Titles 3 and 8; 6 NYCRR Parts 700-706.** Water quality standards are established under various sections of the New York ECL, including Article 15 (ECL § 15-0313) and Article 17 (ECL §§ 17-0301, 17-0303, and 17-0809). The water quality standards for COCs and SOCs are provided in 6 NYCRR § 703.5 and 6 NYCRR Part 703.2 (and also published in NYSDEC's Technical and Operational Guidance Series [TOGS] Memo 1.1.1, Ambient Water Quality Standards and Guidance Values [NYSDEC, 1998; 1999a]).

- **New York State ECL Article 11, Title 5 – Endangered and Threatened Species of Fish and Wildlife – Species of Special Concern; 6 NYCRR Part 182.** The New York State endangered species legislation enacted in 1970 was designed to complement the federal Endangered Species Act (ESA) by authorizing NYSDEC to adopt the federal endangered species list so that prohibitions of possession or sale of federally listed species and products could be enforced by state enforcement agents. The state list can therefore include species that, while plentiful elsewhere, are endangered in New York. The law was amended in 1981 to authorize the adoption of a list of threatened species that would receive protection similar to endangered species. In addition to the threatened species list, NYSDEC also adopted a list of species of special concern, species for which a risk of endangerment has been documented by NYSDEC. The law and regulations restrict activities in areas inhabited by endangered species. The taking of any endangered or threatened species is prohibited, except under a permit or license issued by NYSDEC. The destroying or degrading the habitat of a protected animal likely constitutes a "taking" of that animal under NY ECL § 11-0535.

- **New York State ECL Article 15, Title 5, and Article 17, Title 3; 6 NYCRR Part 608 – Use and Protection of Waters.** These regulations cover excavation and fill of the navigable waters of the state. No person, local public corporation, or interstate authority may excavate from or place fill, either directly or indirectly, in any of the navigable waters of the state or in marshes, estuaries, tidal marshes, and wetlands that are adjacent to and contiguous at any point to any of the navigable waters of the state, and that are inundated at mean high water level or tide, without a permit (6 NYCRR 608.5). In accordance with CERCLA Section 121(e)(1), no federal, state, or local permits are required for remedial action that is conducted entirely on site, although the remedial action must comply with the substantive technical requirements of this statute and associated regulations.

- **New York ECL Article 17, Title 5, 6 NYCRR Part 701.1.** It shall be unlawful for any person, directly or indirectly, to throw, drain, run, or otherwise discharge into such waters organic or inorganic matter that shall cause or contribute to a condition in contravention of applicable standards.

- **New York ECL Article 17, Title 8; 6 NYCRR Part 750-758 – Water Resources Law.** These regulations provide standards for storm water runoff, surface water, and groundwater discharges. In general, they prohibit discharge of any pollutant to the waters of New York without a State Pollutant Discharge Elimination System (SPDES) permit. In accordance with CERCLA Section 121(e)(1), no federal, state, or local permits are required for remedial action that is conducted entirely on site, although the remedial action must comply with the substantive requirements of the Water Resources Law.

- **New York ECL Article 24 Title 7, Freshwater Wetlands; 6 NYCRR Parts 662 – 665.** Freshwater wetlands of New York State are protected under Article 24 of the ECL, commonly known as the Freshwater Wetlands Act (FWA). Wetlands protected under Article 24 are known as New York State regulated wetlands. The regulated area includes the wetlands themselves and a protective buffer or adjacent area that extends 100 feet landward of the wetland boundary. All freshwater wetlands with an area of 12.4 acres or greater are depicted on a set of maps published by NYSDEC. Wetlands less than 12.4 acres may also be mapped if they have unusual local importance. Four classes of wetlands (Class I, the most valuable, through Class IV, the least valuable) have been established and are ranked according to their ability to perform wetland functions and provide wetland benefits. Vegetative cover, ecological associations, special features, hydrological and pollution control features, distribution, and location are factors considered in the determination of wetland benefit.

- **New York State ECL Article 27, Title 13; 6 NYCRR Part 375 – Inactive Hazardous Waste Disposal Sites.** These regulations establish requirements for the development and implementation of inactive hazardous waste disposal site remedial programs.

3.4.2 Federal Laws and Regulations

- **Federal Water Pollution Control Act, as amended by the Clean Water Act (CWA) – 33 USC § 1251 et seq.; 40 CFR Part 129**. The Federal Water Pollution Control Act provides the authority for USEPA to establish water quality criteria. The toxic pollutant effluent standards are promulgated at 40 CFR 129. The ambient water criterion for COCs in navigable waters are established in 40 CFR § 129.105(a)(4).

- **Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) 42 USC § 103.** The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), commonly known as Superfund, provides authority for USEPA to respond directly to releases or threatened releases of hazardous substances that may endanger public health or the environment. CERCLA established prohibitions and requirements concerning closed and abandoned hazardous waste sites; provided for liability of persons responsible for releases of hazardous waste at these sites; and established a trust fund to provide for cleanup when no responsible party could be identified. The law authorizes two kinds of response actions: 1) short-term removals, where actions may be taken to address releases or threatened releases requiring prompt response; and 2) long-term remedial response actions, that permanently and significantly reduce the dangers associated with releases or threats of releases of hazardous substances that are serious, but not immediately life threatening. These actions can be conducted only at sites listed on USEPA's NPL. CERCLA was amended by the Superfund Amendments and Reauthorization Act (SARA) in 1986.

- **40 CFR Parts 9, 122, 123, 131, and 132, Tuesday March 23, 1995, Final Water Quality Guidance for the Great Lakes System Final Rule**. The Guidance consists of water quality criteria for 29 pollutants to protect aquatic life, wildlife, and human health, and detailed methodologies to develop criteria for additional pollutants.

- **Section 404 of the CWA (Federal Water Pollution Control Act, as amended), 33 USC § 1344; 33 CFR Parts 320 to 329.** Section 404 of the CWA establishes requirements for issuing permits for the discharge of dredged or fill material into navigable waters of the United States, and includes special policies, practices, and procedures to be followed by the US Army Corps of Engineers (USACE) in connection with the review of applications for such permits. These regulations apply to all existing, proposed, or potential disposal sites for discharges of dredged or fill materials into US waters, including wetlands. USEPA may prohibit fill if there is an unacceptable adverse impact on the receiving water body. In accordance with CERCLA Section 121(e)(1), no federal, state, or local permits are required for remedial action conducted entirely on site, although the remedial action must comply with the substantive requirements of CWA Sections 404 and 33 CFR Parts 320 to 329.

- **CWA Section 404 (33 USC § 1344), 40 CFR Part 230.** No activity that adversely affects an aquatic ecosystem (including wetlands) shall be permitted if there is a practical alternative available that has less adverse impact. If there is no practicable alternative, then the adverse impacts of the activity must be minimized.

- **Statement of Procedures on Floodplain Management and Wetlands Protection; 40 CFR Part 6, Appendix A.** These procedures set forth USEPA policy and guidance for carrying out Executive Orders (EO) 11990 and 11988.

  - **EO 11988 – Floodplain Management.** Requires federal agencies to evaluate the potential effects of actions that may be taken in a floodplain and to avoid, to the extent possible, long-term and short-term adverse affects associated with the occupancy and modification of floodplains, and to avoid direct or indirect support of floodplain development wherever there is a practicable alternative.

  - **EO 11990 – Protection of Wetlands.** Requires that activities conducted by federal agencies avoid, to the extent possible, long-term and short-term adverse affects associated with the modification or destruction of wetlands. Federal agencies are also required to avoid direct or indirect support of new construction in wetlands when there are practical alternatives; harm to wetlands must be minimized when there is no practical alternative available.

- **Endangered Species Act, 16 USC§ 1531 et seq.; 50 CFR Parts 17, Subpart I, and 50 CFR Part 402.** The ESA of 1973 and subsequent amendments provide for the conservation of threatened and endangered species of animals and plants, and the habitats in which they are found. The act requires federal agencies, in consultation with the Secretary of Interior, to verify that any action is not likely to jeopardize the continued existence of any endangered or threatened species or its critical habitat, or result in the destruction or adverse modification of a critical habitat of such species. Exemptions may be granted by the Endangered Species Committee.

- **Fish and Wildlife Coordination Act, USFWS, 16 USC 661 – 667e.** Whenever the waters of any stream or other body of water are proposed or authorized to be impounded, diverted, the channel deepened, or the stream or other body of water otherwise controlled or modified for any purpose, by any department or agency of the United States, such department or agency first shall consult with the United States Fish and Wildlife Service, Department of the Interior, and with the head of the agency exercising administration over the wildlife resources of the particular state in which the impoundment, diversion, or other control facility is to be constructed, with a view to the conservation of wildlife resources by preventing loss of and damage to such resources.

- **USEPA. Quality Criteria for Water.** This USEPA document (USEPA, 1986a) provides water quality criteria for the effects on freshwater species of organic and inorganic contaminants.

- **USEPA. Update #1 to Quality Criteria for Water.** This USEPA (1986b) document provides water quality criteria for the effects on freshwater species of organic and inorganic contaminants .

- **USEPA. Update #2 to Quality Criteria for Water.** This USEPA (1987b) document provides water quality criteria for the effects on freshwater species of organic and inorganic contaminants.

- **USEPA. Quality Criteria for Water, Update.** This USEPA (1991) document provides water quality criteria for the effects on freshwater species of organic and inorganic contaminants.

- **USEPA. National Recommended Water Quality Criteria – Correction.** This USEPA (1999c) document provides water quality criteria for the effects on freshwater species of organic and inorganic contaminants.

3.4.3 State and Federal Guidance

- **NYSDEC Fish and Wildlife Impact Analysis for Inactive Hazardous Waste Sites.** This report (NYSDEC, 1994a) provides guidance for evaluating ecological impacts in areas contaminated with hazardous materials.

- **NYSDEC Freshwater Wetlands Delineation Manual.** This document (NYSDEC, 1995) provides the technical requirements for wetlands delineation in New York State.

- **NYSDEC Freshwater Wetlands Regulations Guidelines on Compensatory Mitigation.** This document (NYSDEC, 1997c) provides the technical requirements for wetlands mitigation for impacted wetlands in New York State.

- **NYSDEC Division of Fish and Wildlife – Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife, Technical Report 87-3.** This report (Newell et al., 1987) provides a method for calculating contaminant concentration criteria in fish flesh for the protection of piscivorous wildlife, and establishes fish-flesh criteria for various contaminants, including mercury and PCBs.

- **NYSDEC Division of Fish, Wildlife and Marine Resources - Technical Guidance for Screening Contaminated Sediment.** This document (NYSDEC, 1999b) provides sediment screening values for metals and non-polar organic contaminants, such as mercury, PCBs, dioxin/furans, in units of micrograms of contaminant per gram organic carbon in sediment (μg/gOC) for organics and mg/kg dry weight for inorganics.

- **Ecological Risk Assessment Guidance for Superfund (ERAGS), Process for Designing and Conducting Risk Assessments.** This report (USEPA, 1997a) provides guidance on how to design and conduct consistent and technically defensible ecological risk assessments for the Superfund program.

- **USEPA Region 4. Waste Management Division Soil Screening Values for Hazardous Waste Sites.** USEPA Region 4 (1999) provides soil screening values for organic and inorganic contaminants effects on terrestrial species.

- **USEPA. Consensus-Based Freshwater Sediment Quality Guidelines.** This document (Ingersoll et al., 2000) evaluates the ability of consensus-based probable effect concentrations (PECs) to predict sediment toxicity building on the work of MacDonald et al. (2000).

- **USEPA. Ecotox Thresholds.** The Ecotox thresholds (USEPA, 1996a) provide water and sediment quality values for organic and inorganic contaminants effects on aquatic species for use in ecological risk assessments at Superfund sites.

- **USEPA. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod *Hyalella azteca* and the Midge *Chironomus riparius*.** This report (USEPA, 1996b) provides toxicological benchmarks for contaminant effects on sediment-dwelling benthic invertebrates.

- **USEPA. Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning.** This document (USEPA, 1993a) provides methodology and sediment effects guidelines for non-polar organics for the protection of aquatic species.

- **USEPA. Considering Wetlands at CERCLA Sites.** This document (USEPA, 1993c) provides guidance on the methods required at Superfund sites when wetlands are either impacted by contamination or potentially impacted by future remedial efforts.

3.4.4 Other Applicable Guidance

- **Oak Ridge National Laboratory (ORNL) Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision.** This report (Efroymson et al., 1997a) provides toxicological benchmarks for contaminant effects on terrestrial plants.

- **ORNL Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision.** This report (Efroymson et al., 1997b) provides toxicological benchmarks for contaminant effects on terrestrial invertebrates.

- **ORNL Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision.** This report (Jones et al., 1997) provides toxicological benchmarks for contaminant effects on sediment-dwelling benthic invertebrates.

- **Long et al. 1995. Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments.** This study (Long et al., 1995) provides toxicological benchmarks for contaminant effects on sediment-dwelling benthic invertebrates.

- **Persaud, et al. 1993. Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario. Ontario Ministry of the Environment.** This report (Persaud et al., 1993) provides sediment quality values for organic and inorganic contaminants effects on sediment-dwelling benthic invertebrates.

- **ORNL Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision.** This report (Suter and Tsao, 1996) provides toxicological benchmarks for contaminant effects on aquatic species.

- **ORNL Toxicological Benchmarks for Wildlife: 1996 Revision.** This report (Sample et al., 1996) provides toxicological benchmarks for contaminant effects on birds and mammals.

- **Washington State Department of Ecology. Creation and Analysis of Freshwater Sediment Quality Values in Washington State.** This report (Washington State Department of Ecology [WSDE],1997) provides sediment quality values for organic and inorganic contaminants effects on freshwater sediment-dwelling benthic invertebrates.

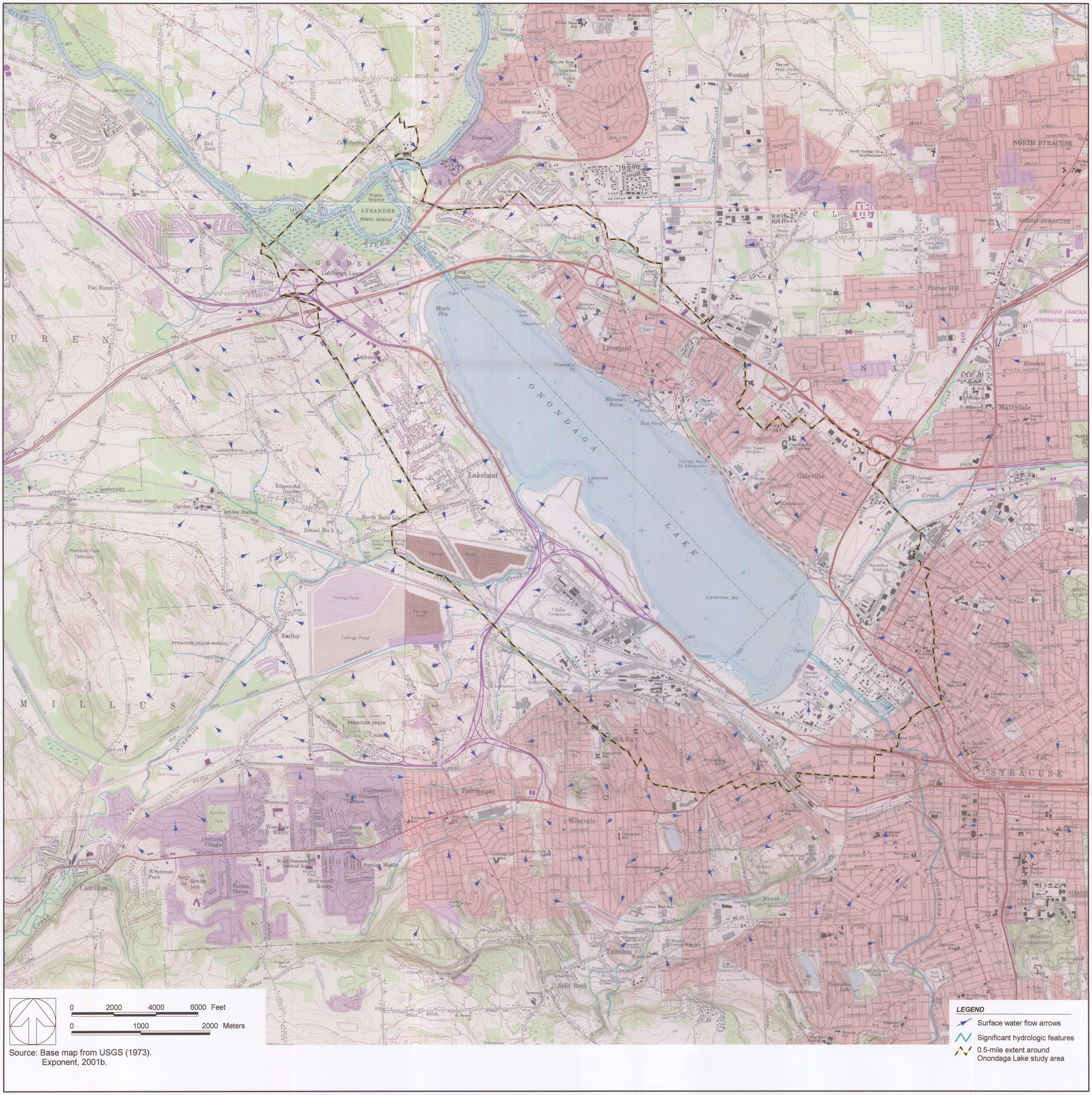


Figure 3-2. Surface Water Flow Patterns in the Onondaga Lake Area

LEGEND

NYSDEC Regulated Wetlands

NWI wetlands

- L1 Lacustrine (limnetic)
- L2 Lacustrine (littoral)
- PEM Palustrine (emergent)
- PFO Palustrine (forested)
- PSS Palustrine (scrub-shrub)
- PUB Palustrine (unconsolidated bottom)
- R2 Riverine (low perennial)

Two-mile (3.2-km) extent around Onondaga Lake study area

Natural Heritage Program Sensitive Areas

RarePlant

1 Sartwell's sedge
2 Little-leaf tick-trefoil
3 Red pigweed

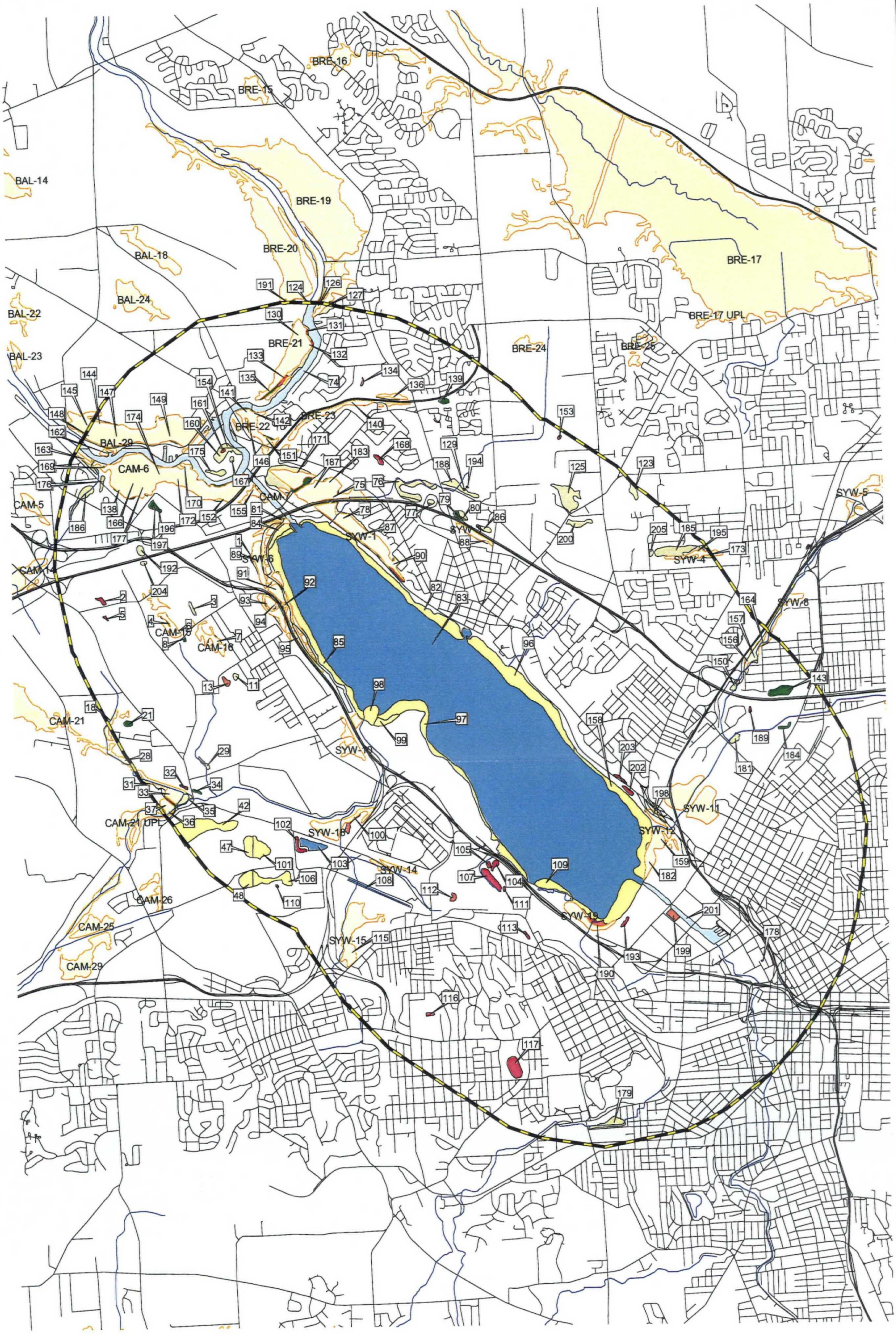
Inland salt pond

0 2000 4000 6000 Feet

0 1000 2000 Meters

Source: Base map from USGS (1973)
Modified from Exponent, 2001b.

Figure 3-1. Topographic Features of Onondaga Lake and Vicinity



LEGEND

NWI wetlands attributes

- L1 Lacustrine (limnetic)
- L2 Lacustrine (littoral)
- PEM Palustrine (emergent)
- PFO Palustrine (forested)
- PSS Palustrine (scrub-shrub)
- PUB Palustrine (unconsolidated bottom)
- R2 Riverine (low perennial)

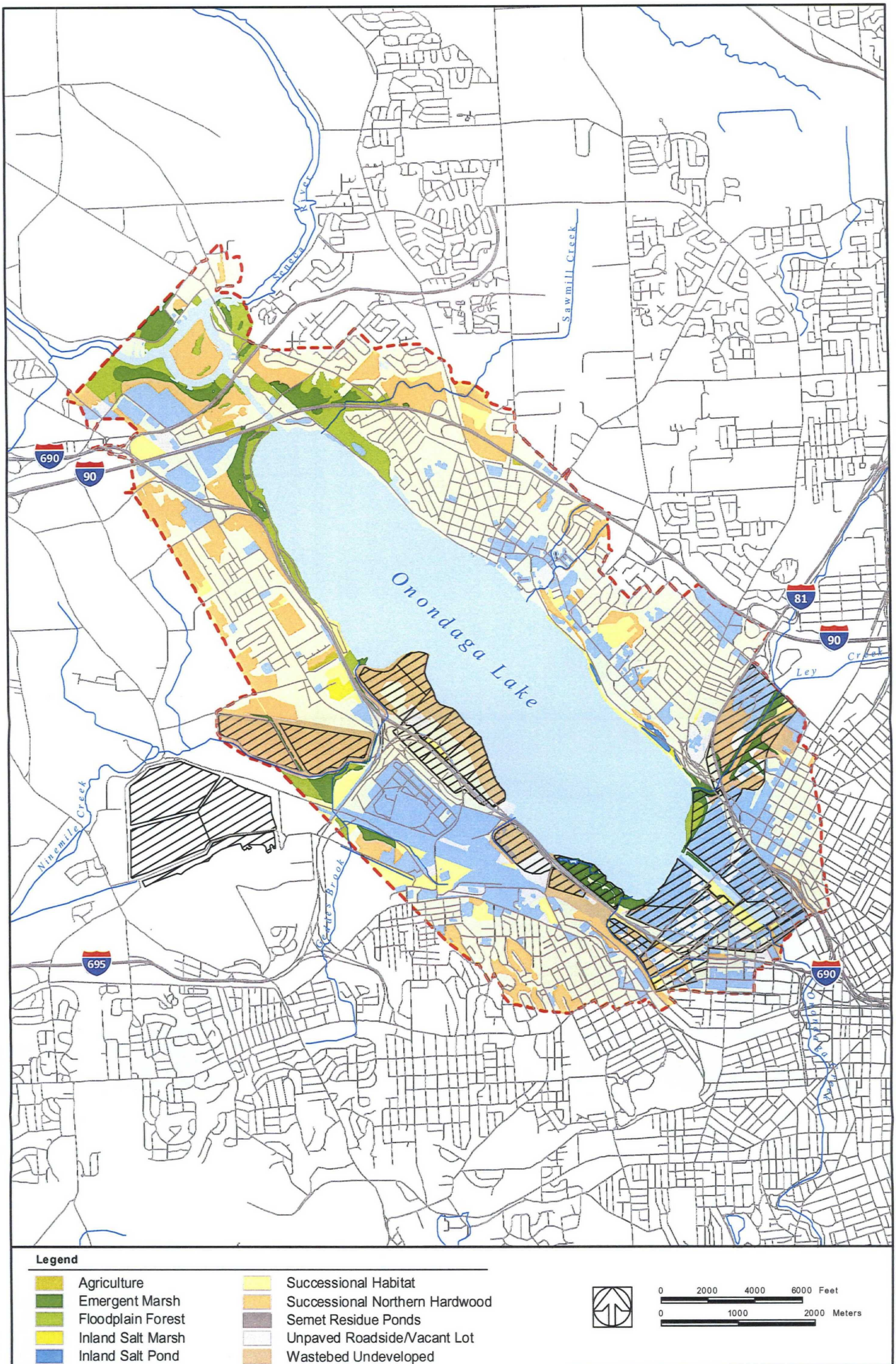
- NYSDEC wetlands
- Two-mile (3.2-km) extent around Onondaga Lake study area
- 3 Sequential number corresponding to BERA Table 3-3
- SYW-14 NYSDEC wetland identification code

0 3000 6000 Feet

0 1000 2000 Meters

Source: Base map from NYSDOT (no date)
Exponent, 2001b.

Figure 3-3. Wetlands Surrounding Onondaga Lake Study Area



Source: Base map from NYSDOT (no date). Covertypes near lake shore confirmed by NYSDEC, 2001. Modified from Exponent, 2001b.

Figure 3-4. Covertypes within 0.5 mile (0.8 km) of the Onondaga Lake Study Area

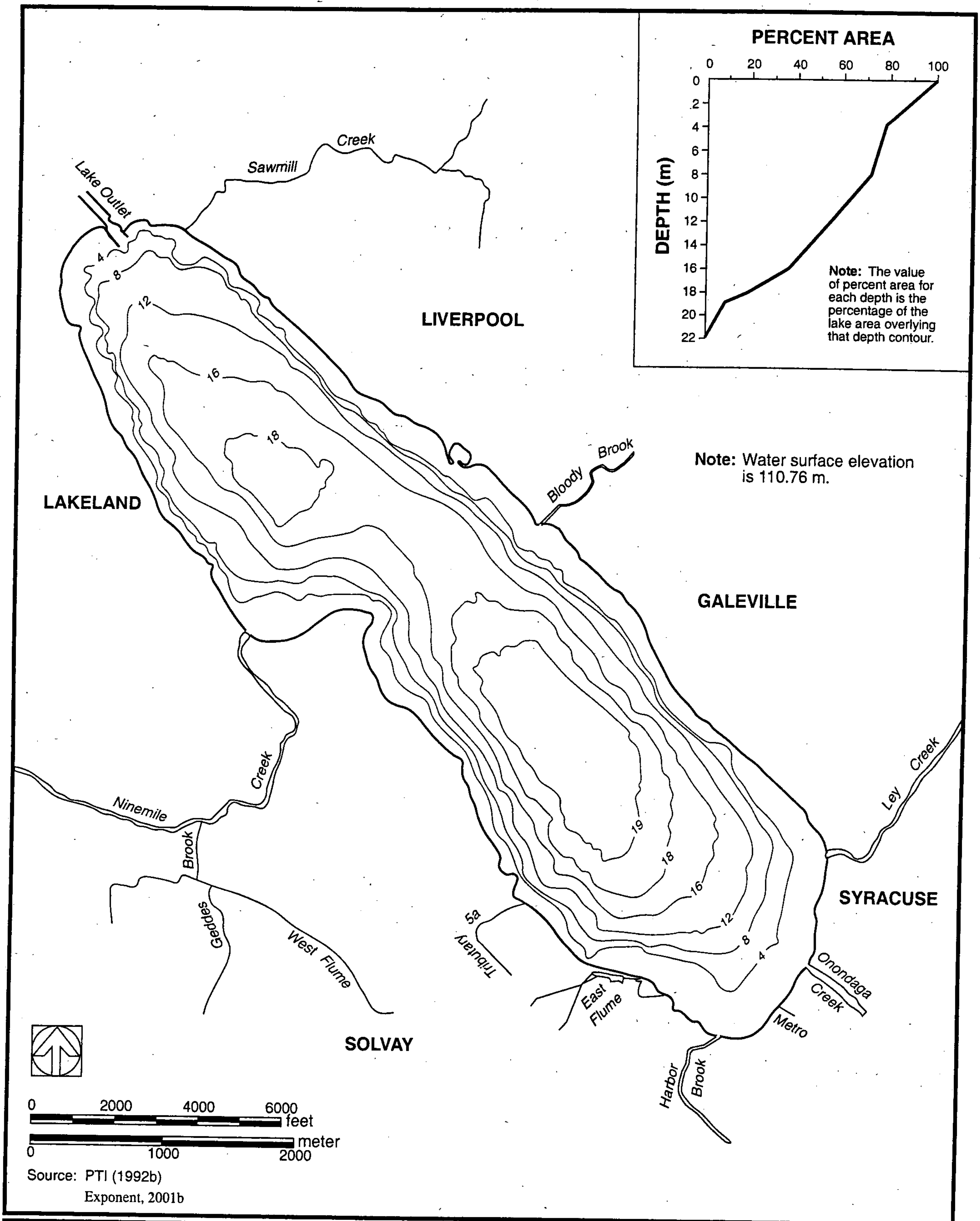
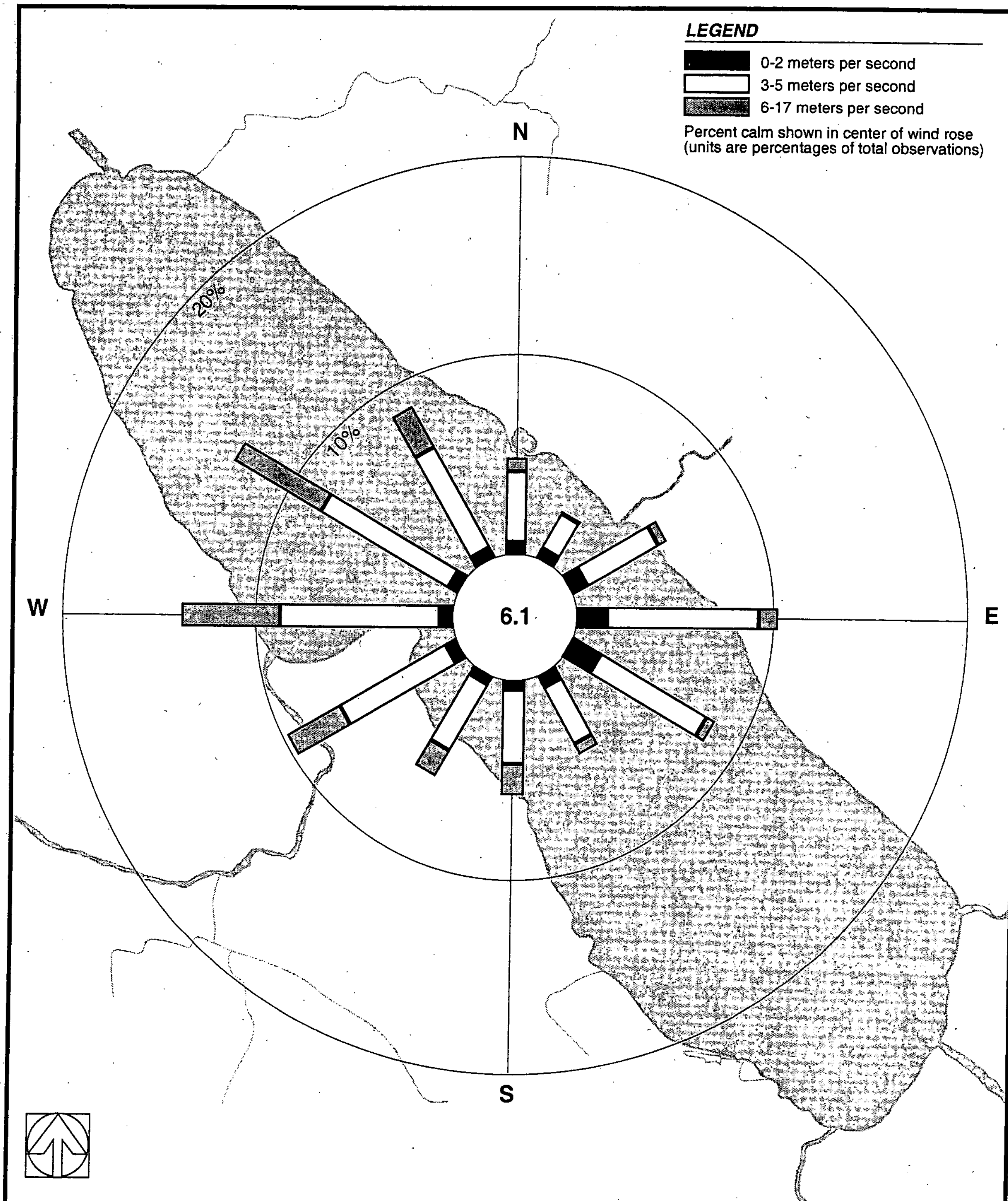


Figure 3-5. Bathymetry and hypsographic curve (inset) for Onondaga Lake



Source: National Oceanographic and Atmospheric Administration hourly observations at Hancock International Airport, Syracuse, New York
Exponent, 2001b

Figure 3-6. Annual wind rose for Onondaga Lake during 1983 to 1992

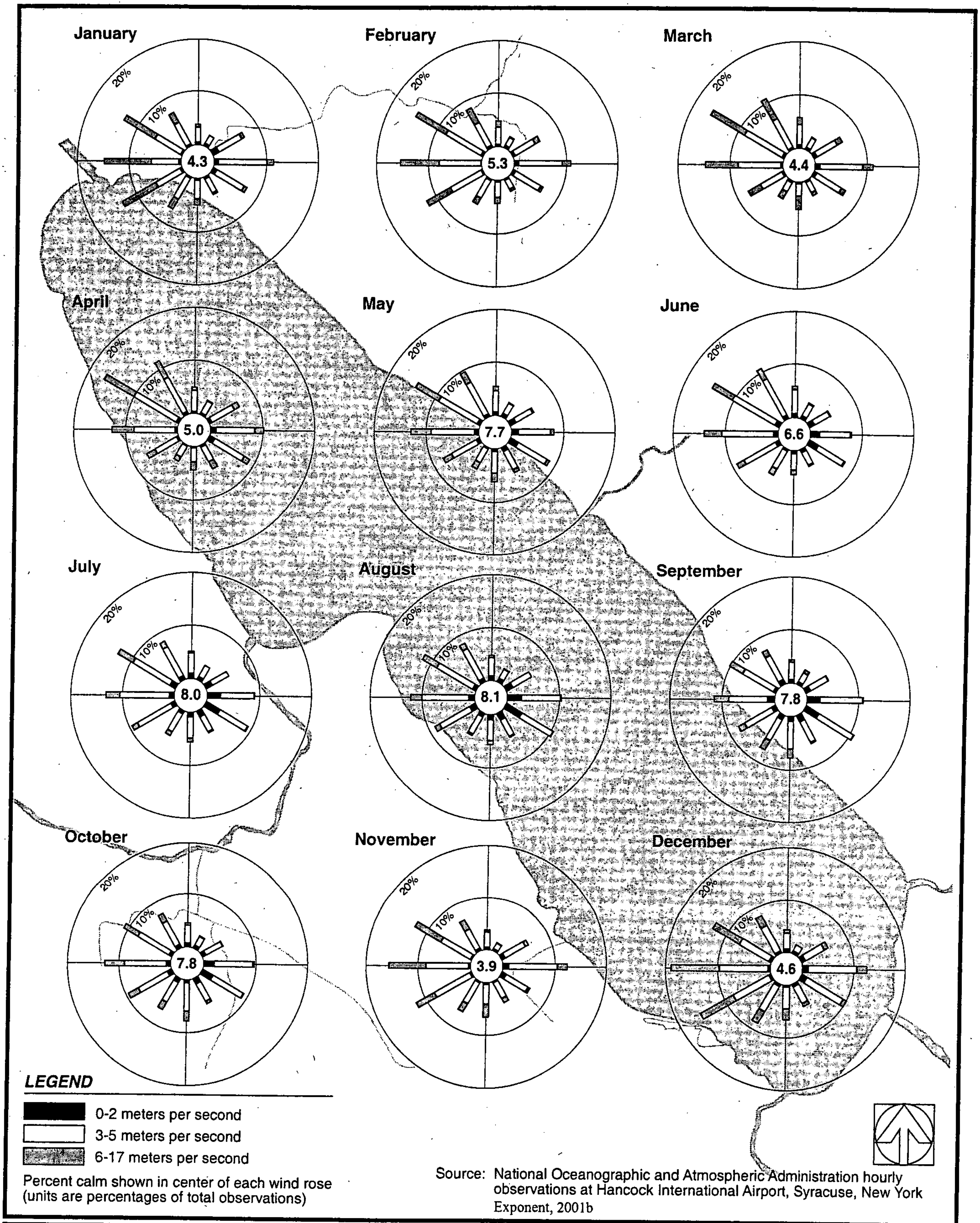
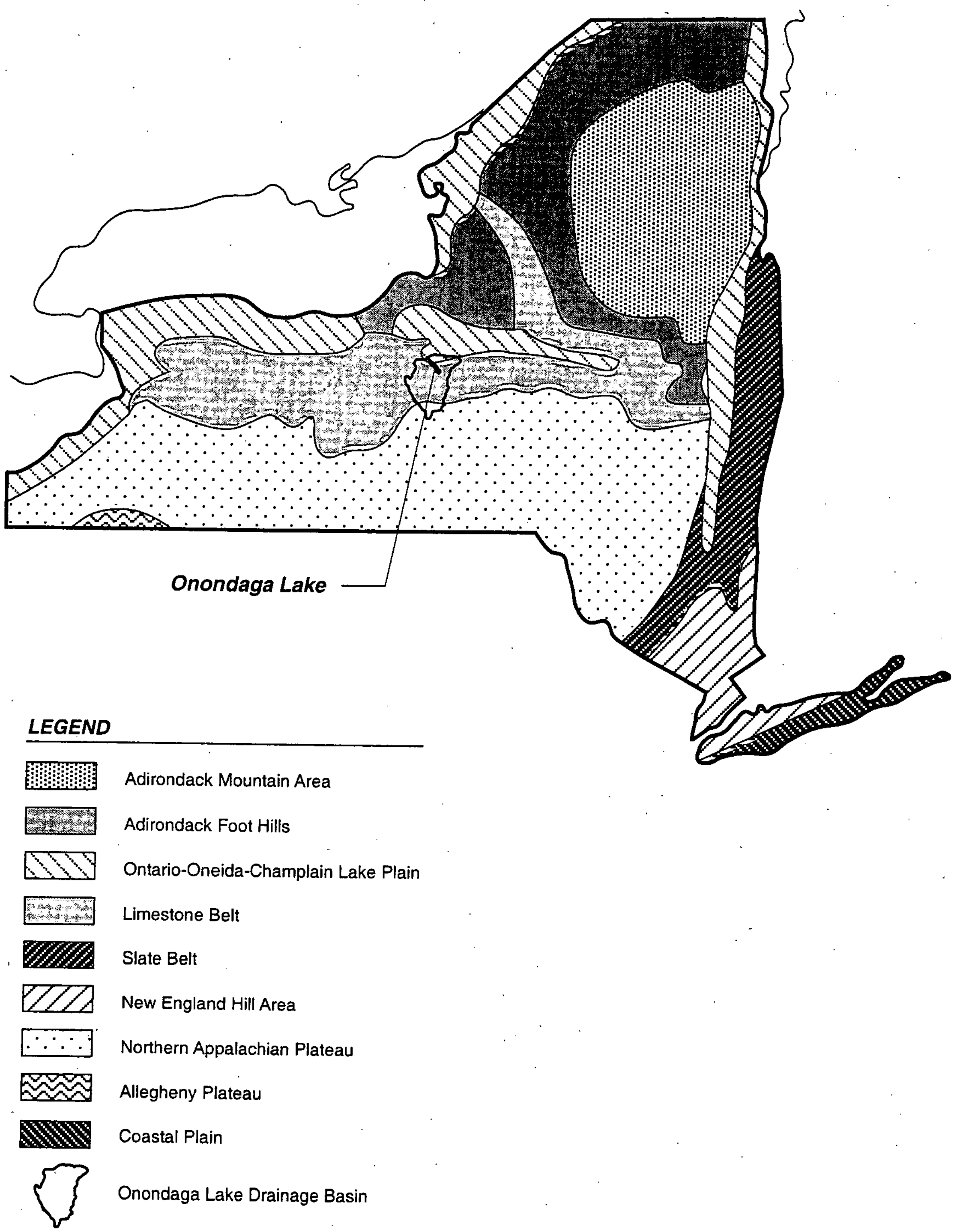
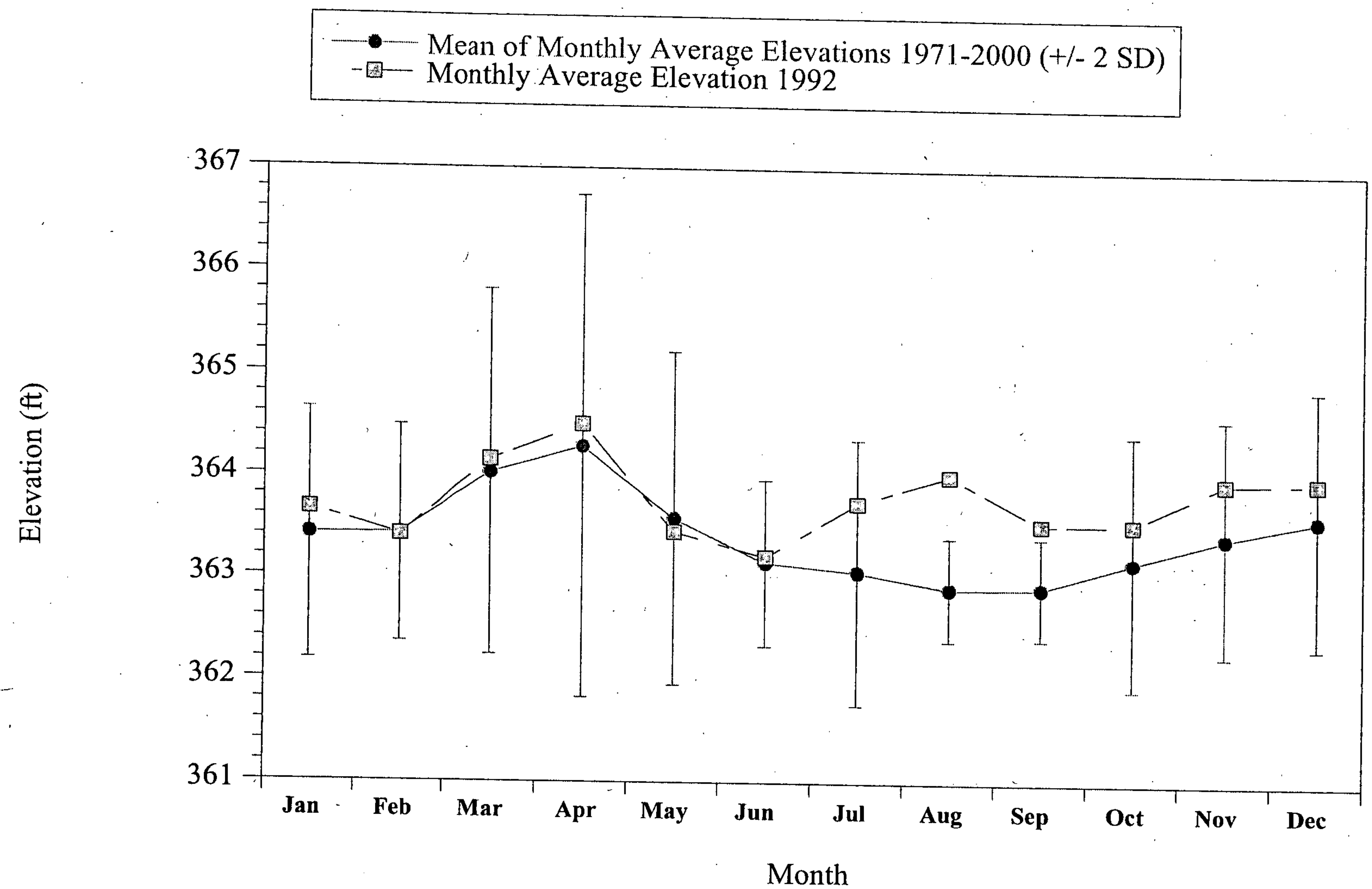


Figure 3-7. Monthly wind roses for Onondaga Lake during 1983 to 1992



Source: Berg (1963)
Exponent, 2001b

Figure 3-8. Physiographic regions of New York State



Reference: USGS, 2001

Figure 3-9
Monthly Average Elevations of Onondaga Lake

TAMS

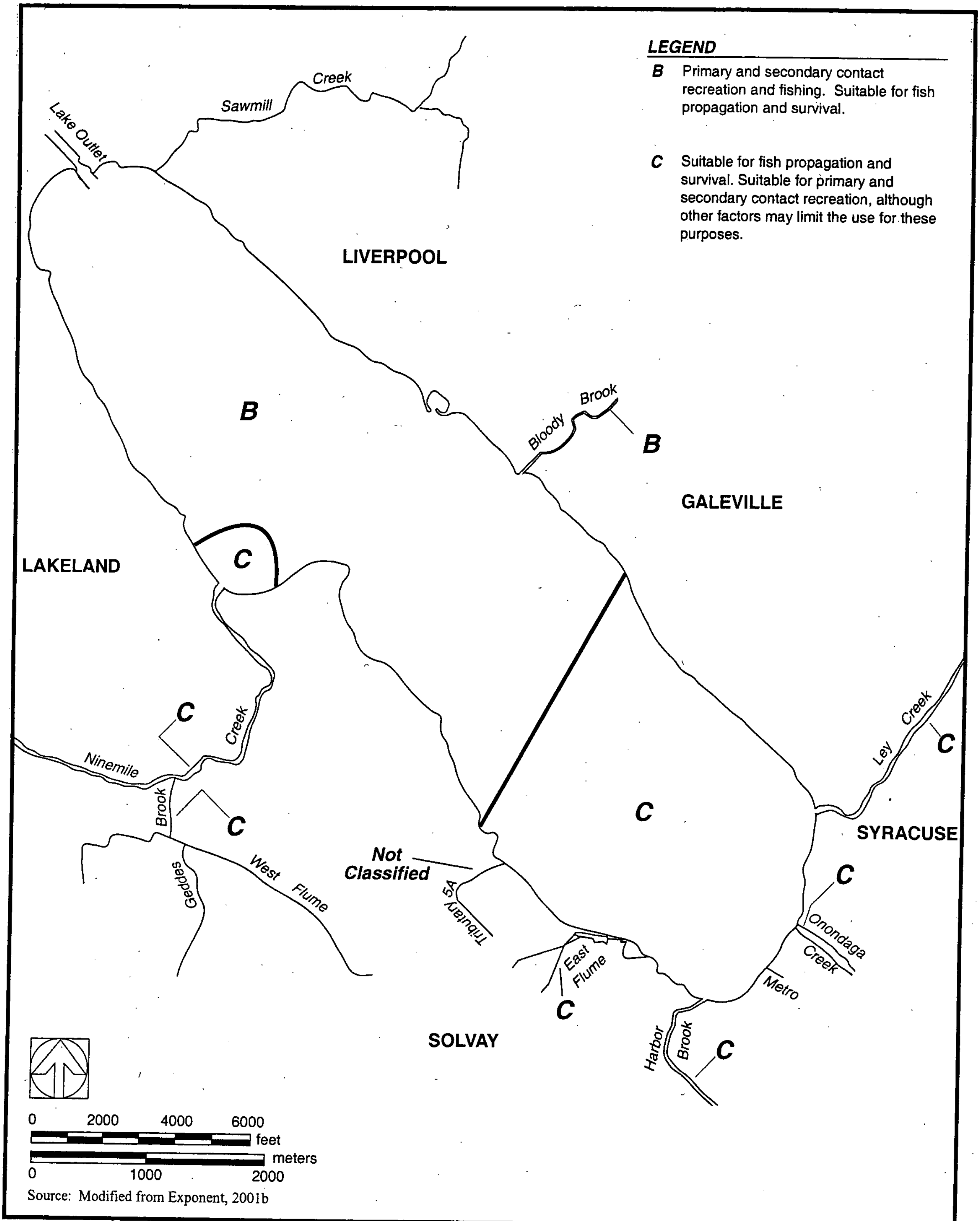
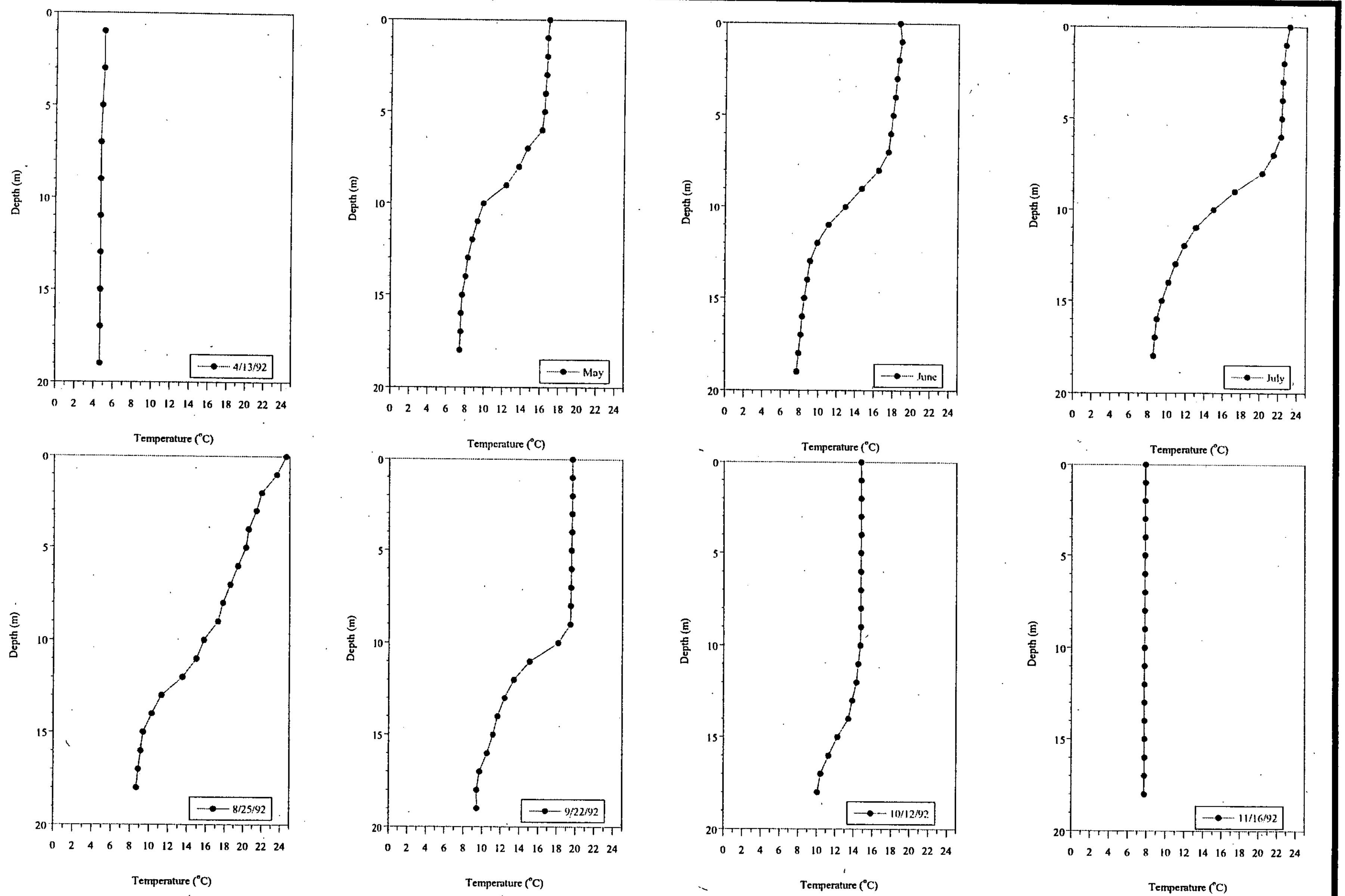


Figure 3-10. New York State Water Quality Classifications within and around Onondaga Lake



Note: The profiles shown for May, June and July are the averages for two readings within the month, therefore, specific dates are not shown.

TAMS

Figure 3-11
Monthly Temperature Profiles in South Basin 1992

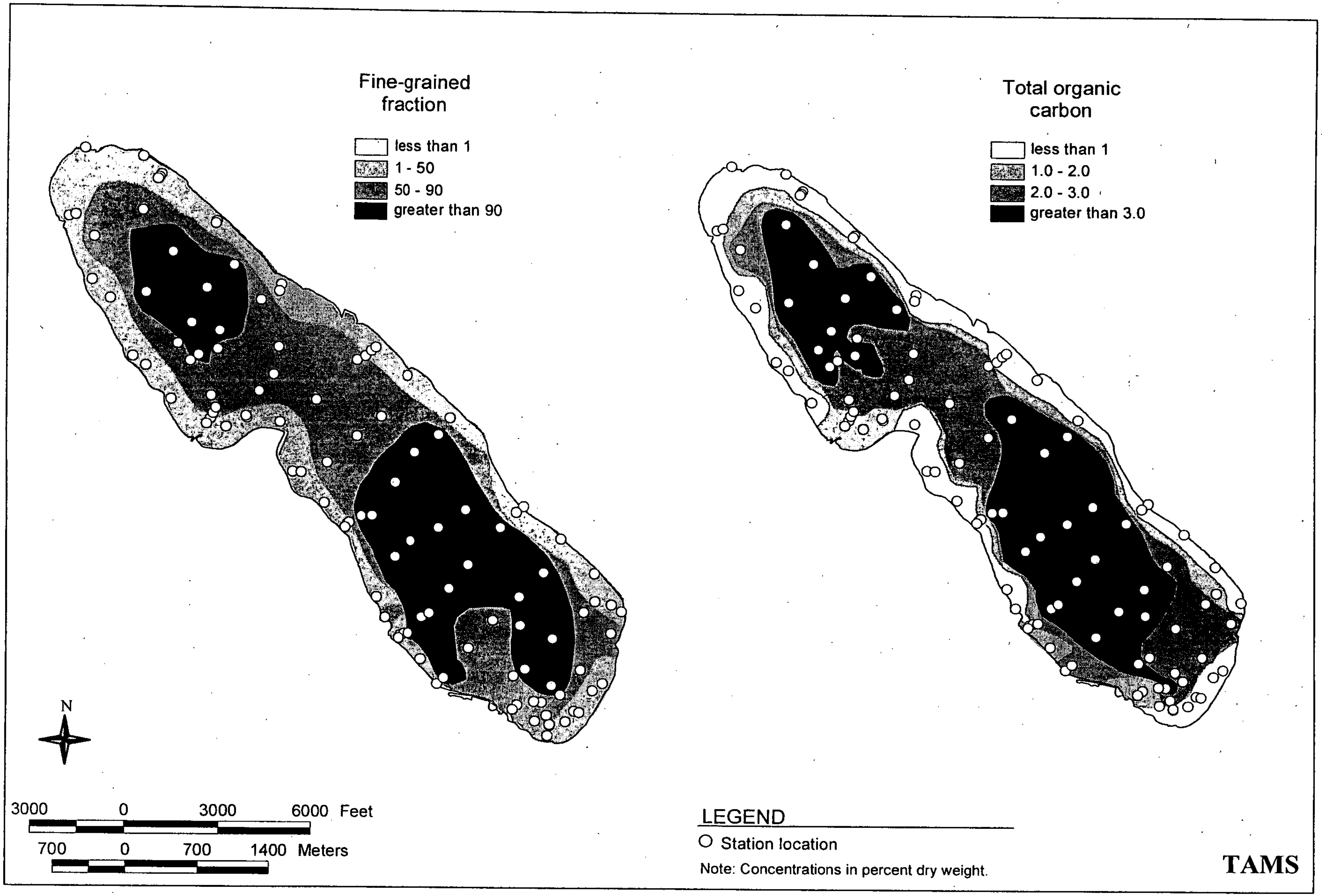
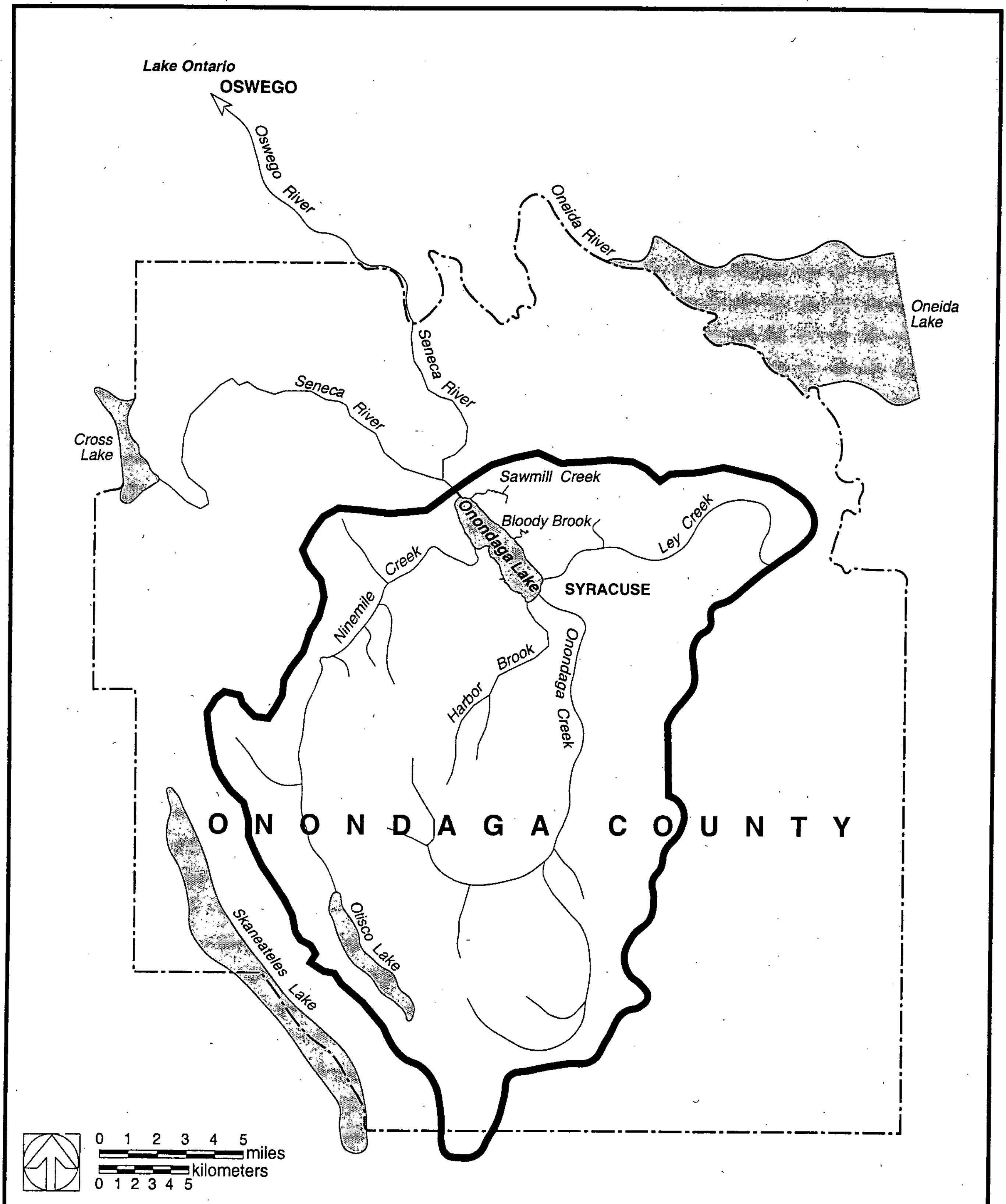
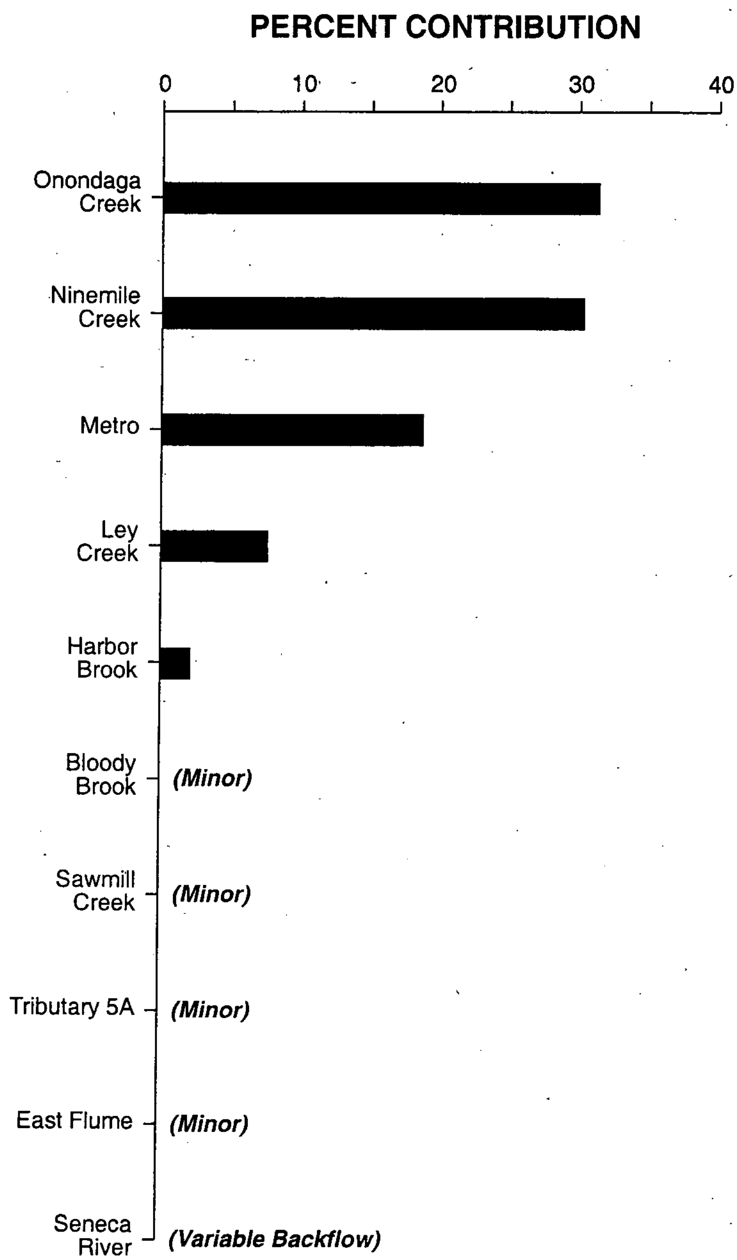


Figure 3-12. Fine-grained Fraction and Total Organic Carbon Content of Surficial Sediments of Onondaga Lake in 1992



Source: Onondaga County (1971)
Exponent, 2001b

Figure 3-13. Onondaga Lake drainage basin



Source: Effler and Whitehead (1996)
Modified from Exponent, 2001b

Figure 3-14. Relative Contribution of Tributaries and Metro to Total Inflow to Onondaga Lake from 1971 to 1989

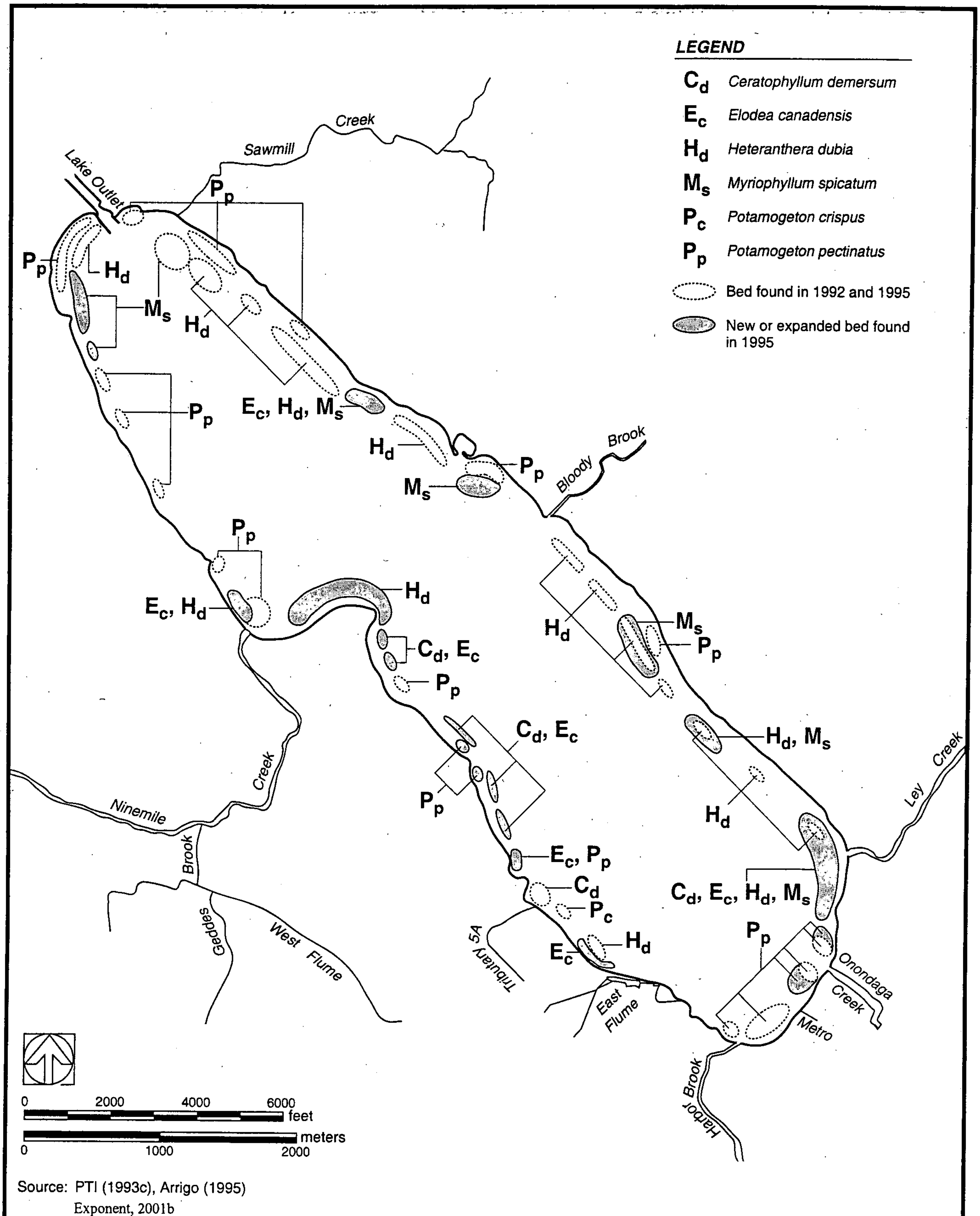


Figure 3-15. Distributions of major macrophyte beds in Onondaga Lake in 1992 and 1995

Table 3-1. Minimum and Maximum Elevations of Onondaga Lake for the 10-Year Period 1983 to 1992[a]

Year	Elevation Minimum	Elevation Maximum	Difference in Elevation (ft)
1983	362.4	367.1	4.7
1984	362.4	365.8	3.4
1985	362.5	365.6	3.1
1986	362.0	365.7	3.7
1987	362.1	364.4	2.3
1988	362.4	363.9	1.5
1989	362.2	366.1	3.9
1990	362.5	365.8	3.3
1991	362.5	365.8	3.3
1992	362.6	365.8	3.2

Source: USGS unpublished data records.

[a] Elevations are expressed in feet above sea level.

Table 3-2. Characteristics of NYSDEC-regulated Wetlands Within 2 miles (3.2 km) of Onondaga Lake

Wetland Identification Code	Class	Area (ha)	Predominant Vegetation
SYW-1	I	54.4	Deciduous trees and shrubs mixed with emergent vegetation
SYW-3	II	13.4	Deciduous trees and shrubs mixed with emergent vegetation; pockets of meadow vegetation present
SYW-4	III	5.3	Deciduous trees
SYW-6	I	40.6	Emergent vegetation and deciduous shrubs dominate; living and dead deciduous trees and floating vegetation also present
SYW-8	II	13.0	Deciduous trees and shrubs and emergent vegetation
SYW-10	I	11.0	Deciduous trees and shrubs and emergent vegetation
SYW-11	II	16.6	Reeds (*Phragmites*)
SYW-12	I	16.5	Reeds are dominant, some deciduous trees and shrubs also present
SYW-14	III	5.0	Emergent vegetation; herbaceous and shrubby successional vegetation
SYW-15	II	20.7	Emergent vegetation dominates with wet-meadow vegetation, dead trees, living deciduous shrubs, and a few living deciduous trees
SYW-18	II	11.0	Reeds are dominant[a]
SYW-19	II	8.0	Reeds are dominant[a]
BAL-29	II	68.9	Deciduous trees and shrubs and emergent vegetation dominate; wet meadow and upland vegetation also present

Table 3-2.(cont.)

Wetland Identification Code	Class	Area (ha)	Predominant Vegetation
BRE-19	II	144.76	Deciduous trees and shrubs; emergent and wet meadow vegetation
BRE-21	II	27.4	Emergent vegetation, deciduous trees and shrubs, and upland vegetation
BRE-22	I	12.6	Deciduous trees and shrubs, wet meadow vegetation, and emergent vegetation
BRE-23	II	16.8	Emergent vegetation dominates; deciduous trees, shrubs, and some dead trees also present
CAM-6	I	70.7	Deciduous trees and emergent vegetation, with some deciduous shrubs and wet meadow vegetation
CAM-7	II	5.6	Deciduous trees, emergent and floating vegetation
CAM-15	III	8.3	Deciduous shrubs are dominant; wet meadow vegetation, dead/living deciduous trees present
CAM-16	II	8.4	Wet meadow vegetation and deciduous trees and shrubs
CAM-21	II	126.7	Deciduous trees and shrubs, dead trees, emergent vegetation, and wet-meadow vegetation

Sources: Rhodes and Alexander (1980)

[a] Monastory (1995, pers. comm.)

Table 3-3. Attributes of NWI Wetlands Within 2 Miles (3.2 km) of Onondaga Lake

Number in Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
1	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
2	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Excavated
3	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
4	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
5	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Excavated
6	PSS1F	Palustrine	Scrub-shrub	Broad-leaved deciduous	Semipermanently flooded	–
7	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
8	PSS1F	Palustrine	Scrub-shrub	Broad-leaved deciduous	Semipermanently flooded	–
11	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
13	PEM5F	Palustrine	Emergent	[a]	Semipermanently flooded	–
18	PEM5C	Palustrine	Emergent	[a]	Seasonally flooded	–
21	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
28	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
29	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
31	PFO1/SS1Cd	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded	Partly drained/ ditched
32	PUBF	Palustrine	Unconsolidated bottom	–	Semipermanently flooded	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
33	PSS1/EM5A	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Temporarily flooded	–
34	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
35	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
36	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
37	PEM/UBF	Palustrine	Emergent/ unconsolidated bottom	–	Semipermanently flooded	–
42	L2UBKFhs	Lacustrine	Littoral/unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
47	L2UBKFhs	Lacustrine	Littoral/unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
48	L2UBKFhs	Lacustrine	Littoral/unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
74	R2UBH	Riverine	Low perennial/ unconsolidated bottom	–	Permanently flooded	–
75	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
76	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
77	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
78	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
79	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
80	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
81	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
82	L2UBH	Lacustrine	Littoral/unconsolidated bottom	–	Permanently flooded	–
83	L1UBH	Lacustrine	Limnetic/unconsolidated bottom	–	Permanently flooded	–
84	L2UBG	Lacustrine	Littoral/Unconsolidated bottom	–	Intermittently exposed	–
85	L2UBH	Lacustrine	Littoral/Unconsolidated bottom	–	Permanently flooded	–
86	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
87	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
88	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
89	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
90	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Excavated
91	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
92	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
93	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
94	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
95	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
96	L2UBH	Lacustrine	Littoral	–	Permanently flooded	–
97	L2UBH	Lacustrine	Littoral	–	Permanently flooded	–
98	L2USCs	Lacustrine	Littoral	–	Seasonally flooded	Spoil

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
99	L2UBG	Lacustrine	Littoral	–	Intermittently exposed	–
100	PEM1E	Palustrine	Emergent	Persistent	Seasonally flooded/saturated	–
101	L2UBKFhs	Lacustrine	Littoral/unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
102	PUBKZh	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
103	L1UBKZh	Lacustrine	Limnetic	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
104	PUBKFhs	Palustrine	Unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
105	PUBKZhs	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded, spoil
106	L2UBKFhs	Lacustrine	Littoral	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
107	PUBKFhs	Palustrine	Unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil
108	L1UBZx	Lacustrine	Limnetic/unconsolidated bottom	–	Intermittently exposed/permanent	–
109	L2UBH	Lacustrine	Littoral/unconsolidated bottom	–	Permanently flooded	–
110	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Excavated
111	PUBKFhs	Palustrine	Unconsolidated bottom	–	Artificially flooded, semipermanently flooded	Diked/ impounded, spoil

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
112	PEM1E	Palustrine	Emergent	Persistent	Seasonally flooded/saturated	–
113	PUBKZh	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
115	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
116	PUBKZh	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
117	PUBKZh	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
123	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
124	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
125	PFO1/SS1C	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded	–
126	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
127	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
129	PUBZh	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Diked/ impounded
130	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
131	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
132	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
133	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
134	PEM5/UBFh	Palustrine	Emergent/ unconsolidated bottom	[a]	Semipermanently flooded	Diked/ impounded
135	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
136	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
138	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
139	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
140	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
141	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
142	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
143	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
144	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
145	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
146	PSS1/EM5Cd	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded	Partially drained/ ditched
147	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
148	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
149	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
150	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
151	PSS1/EM5Cd	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded	Partially drained/ ditched
152	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
153	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently flooded/permanent	Excavated
154	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
155	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
156	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
157	PFO5/1E	Palustrine	Forested	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
158	L2UBH	Lacustrine	Littoral/unconsolidated bottom	–	Permanently flooded	–
159	PEM1Cs	Palustrine	Emergent	Broad-leaved deciduous	Seasonally flooded	Spoil
160	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
161	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
162	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
163	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
164	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
166	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
167	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
168	PUBFx	Palustrine	Unconsolidated bottom	–	Semipermanently flooded	Excavated
169	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
170	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
171	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
172	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
173	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
174	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
175	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
176	PFO1A	Palustrine	Forested	Broad-leaved deciduous	Temporarily flooded	–
177	PSS1/EM5E	Palustrine	Scrub-shrub/emergent	Broad-leaved deciduous[a]	Seasonally flooded/saturated	–
178	PUBFx	Palustrine	Unconsolidated bottom	–	Semipermanently flooded	Excavated
179	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
180	L2BBAs	Lacustrine	Littoral	[b]	Temporarily flooded	–
181	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
182	PEM1Cs	Palustrine	Emergent	Broad-leaved deciduous	Seasonally flooded	Spoil
183	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
184	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
185	PFO1Cd	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	Partially drained/ ditched
186	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
187	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
188	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
189	PUBZx	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Excavated
190	PEM1Cs	Palustrine	Emergent	Broad-leaved deciduous	Seasonally flooded	Spoil
191	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
192	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
193	PUBKZh	Palustrine	Unconsolidated bottom	–	Artificially flooded, intermittently exposed/permanent	Diked/ impounded
194	PUBZh	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Diked/ impounded
195	PFO1E	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded/saturated	–
196	PSS1E	Palustrine	Scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
197	PEM5E	Palustrine	Emergent	[a]	Seasonally flooded/saturated	–
198	PEM1Cs	Palustrine	Emergent	Broad-leaved deciduous	Seasonally flooded	Spoil
199	PEM1/UBFx	Palustrine	Emergent/ unconsolidated bottom	Broad-leaved deciduous	Semipermanently flooded	Excavated

Table 3-3. (cont.)

Number on Figure 3-3	Attribute	System	Class	Subclass	Water Regime	Special Modifiers
200	PFO1C	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	–
201	R2UBHx	Riverine	Lower perennial/ unconsolidated bottom	–	Permanently flooded	Excavated
202	PUBZh	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Diked/ impounded
203	PUBZh	Palustrine	Unconsolidated bottom	–	Intermittently exposed/permanent	Diked/ impounded
204	PFO1/SS1E	Palustrine	Forested/scrub-shrub	Broad-leaved deciduous	Seasonally flooded/saturated	–
205	PFO1Cd	Palustrine	Forested	Broad-leaved deciduous	Seasonally flooded	Partially drained/ ditched

Source: USFWS (1999)

Notes: [a] No definition is available for subclass "5"

[b] No definition is available for subclass "BB"

Table 3-4. Phytoplankton Taxa Collected in Onondaga Lake in 1992

Species	Species
Green Algae	**Diatoms**
Chlamydomonas spp.	*Melosira granulata*
Chlorogonium sp.	*Coscinodiscus* sp.
Heteromastix angulata	*Cyclotella* spp.
Platymonas elliptica	*Stephanodiscus* spp.
Schroederia setigera	*Diatoma elongatum*
Dictyosphaerium pulchellum	*Diatoma tenue*
Pediastrum duplex	*Fragilaria crotonensis*
Coelastrum microporum	*Synedra* spp.
Chlorella vulgaris	*Asterionella formosa*
Oocystis parva	*Navicula* sp.
Ankistrodesmus falcatus	*Nitzschia palea*
Scenedesmus obliquus	**Dinoflagellates**
Scenedesmus quadricauda	*Ceratium hirundinella*
Kirchneriella elongata	***Cryptomonads***
Quadrigula lacustris	*Chroomonas* sp.
Cruciginia tetrapedia	*Cryptomonas erosa*
Cosmarium sp.	**Blue-Green Algae**
Straurastrum sp.	*Microcystis* sp.
	Anabaena spp.
	Aphanizomenon flos-aquae
	Raphidiopsis sp.

Sources: PTI (1993c); Stearns & Wheler (1994)

Table 3-5. Zooplankton Taxa Collected in Onondaga Lake Between 1986 and 1989

Species	Relative Abundance
Cladocerans	
Bosmina longirostris	C
Ceriodaphnia quadrangula	C
Daphnia galeata	C
Daphnia pulex	C
Diaphanosoma leuchtenbergianum	C
Eubosmina coregoni	R
Leptodora kindtii	R
Copepods	
Cyclops bicuspidatus	R
Cyclops vernalis	C
Diaptomus siciloides	C
Rotifers	
Brachionus angularis	C
Brachionus calyciflorus	C
Brachionus variabilis	C
Filinia longiseta	C
Filinia terminalis	C
Kellicottia bostoniensis	C
Kellicottia longispina	C
Keratella cochlearis	R
Keratella quadrata	C
Keratella robusta	C
Keratella testudo	C
Nothalca squamula	R
Ploesoma truncatum	R
Polyarthra sp.	C
Trichocerca multicrinnus	R

Source: Siegfried et al. (1996)

Note: R - rare
C - common

Table 3-6. Benthic Macroinvertebrate Taxa Collected in Onondaga Lake in 1992 and 2000

Phyllum	Class	Order	Family	Genus/Species
Nematoda				
Platyhelmenthes	Turbellaria	Seriata	Planariidae	*Dugesia*
				Dugesia tigrina
Rhynchocoela				
Annelida	Oligochaeta	Lumbriculida	Lumbriculidae	*Stylodrilus heringianus*
		Oligochaeta (Tubificida)	Naididae	*Dero*
				Dero digitata
				Nais bretscheri
				Nais communis
				Ophidonais serpentina
				Stylaria lacustris
				Vejdovskyella intermedia
			Tubificidae	*Aulodrilus pigueti*
				Ilyodrilus templetoni
				Limnodrilus
				Limnodrilus cervix
				Limnodrilus cervix variant
				Limnodrilus claparedeianus
				Limnodrilus hoffmeisteri
				Limnodrilus profundicola
				Limnodrilus udekemianus
				Potamothrix bavaricus
				Potamothrix moldaviensis
				Quistadrilus multisetosus
				Tubifex tubifex
Mollusca	Bivalvia	Heterodonta	Dreissenidae	*Dreissena polymorpha*
			Sphaeriidae	*Pisidium*
				Pisidium casertanum

Table 3-6. (cont.)

Phyllum	Class	Order	Family	Genus/Species
	Gastropoda	Basommatophora	Physidae	*Physa*
				Physa gyrina
				Physa heterostropha
				Physa sp. B
			Planorbidae	*Gyraulus*
				Gyraulus circumstriatus
		Mesogastropoda	Valvatidae	*Valvata piscinalis*
				Gyraulus parvus
	Pelecypoda	Heterodonta	Dreissenidae	*Dreissina polymorpha*
			Sphaeriidae	*Pisidium compressum*
				Pisidium dubium
				Pisidium walkeri
				Sphaerium
				Sphaerium corneum
				Sphaerium fabale
				Sphaerium nitidum
				Sphaerium patella
				Sphaerium rhomboideum
Arthropoda	Arachnida	Acarina	Sperchontidae	*Sperchon*
				Sperchon sp. B
			Unionicolidae	*Neumania*
				Neumania sp. A
		Amphipoda	Gammaridae	*Gammarus*
		Hydrachnida		
		Trombidiformes	Limnesiidae	*Limnesia*

Table 3-6. (cont.)

Phyllum	Class	Order	Family	Genus/Species
Arthropoda	Crustacea			*Gammarus fasciatus*
				Gammarus pseudolimnaeus
				Gammarus tigrinus
		Diplostraca	Macrothricidae	*Ilyocryptus*
		Isopoda	Asellidae	*Caecidotea*
				Caecidotea racovitzai
		Podocopa	Cypridae	
		Coleoptera	Elmidae	*Dubiraphia*
Arthropoda	Insecta			*Macronychus*
				Stenelmis
			Staphylinidae	
		Collembola	Entomobryidae	*Entomobrya* sp. A
		Diptera	Blephariceridae	
			Ceratopogonidae	
			Chironomidae	*Chironomini-tribe*
				Chironomidae genus AM
				Chironomidae genus BG
				Chironomidae genus S
				Chironomidae genus U
				Chironomus
				Chironomus cf. Riparius
				Chironomus crassicaudaus
				Chironomus decorus grp
				Chironomus plumosus
				Chironomus species A
				Cladopelma
				Cladotanytarsus
				Cricotopus
				Cricotopus sylvestris
				Cryptochironomus

Table 3-6. (cont.)

Phyllum	Class	Order	Family	Genus/Species
				Dicrotendipes
				Dicrotendipes modestus
				Einfeldia
				Endochironomus
				Glyptotendipes
				Labrundinia
				Nanocladius distinctus
				Parachironomus
				Parachironomus carinatus
				Parachironomus directus
				Paratanytarsus
				Polypedilum
				Polypedilum halterale
				Polypedilum simulans group
				Procladius
				Procladius species A
				Procladius-Holotanypus
				Psectrocladius
				Pseudochironomus
				Rheotanytarsus
				Tanypus
				Tanupus stellatus
				Tanytarsus
				Tanytarsus sp. I
			Psychodidae	*Tanytarsus* sp. IV
				Pericoma
				Psychoda
			Tipulidae	*Psychoda alternata*
		Lepidoptera	Pyralidae	
		Odonata	Coenagrionidae	*Acentria*

Sources: PTI (1993c); Exponent 2001data files.

Table 3-7. Fish Species Collected in Onondaga Lake in Selected Years Between 1927 and 1994[a]

		Year Captured								
Common Name	**Species**	**1927**	**1946**	**1969**	**1980**	**1989**	**1990**	**1991**	**1993**	**1994**
Sea lamprey	*Petromyzon marinus*							•		
Gar	*Lepistosteus* sp.				•	•	•	•	•	•
Bowfin	*Amia calva*				•	•	•	•	•	•
Alewife	*Alosa pseudoharengus*		•		•	•		•	•	•
Gizzard shad	*Dorosoma cepedianum*				•	•	•	•	•	•
Rainbow trout	*Oncorhynchus mykiss*							•	•	•
Atlantic salmon	*Salmo salar*								•	•
Brown trout	*Salmo trutta*					•	•	•	•	•
Lake trout	*Salvelinus namaycush*				•					
Brook trout	*Salvelinus fontinalis*							•		
Splake	*Salvelinus* (hybrid)[b]						•			
Trout-perch	*Percopsis omiscomaycus*									•
Rainbow smelt	*Osmerus mordax*							•		•
Central mudminnow	*Umbra limi*							•	•	•
Northern pike	*Esox lucius*		•		•	•	•	•	•	•
Grass pickerel	*Esox americanus*	•								
Chain pickerel	*Esox niger*							•		•
Muskellunge	*Esox masquinongy*[c]									
Tiger muskellunge	*Esox* (hybrid)					•	•	•	•	•
Carp	*Cyprinus carpio*	•	•	•	•	•	•	•	•	•
Golden shiner	*Notemigonus crysoleucas*	•	•		•	•	•	•	•	•
Emerald shiner	*Notropis atherinoides*		•	•		•		•	•	•
Spottail shiner	*Notropis hudsonius*								•	•
Spotfin shiner	*Notropis spilopterus*							•	•	
Redfin shiner	*Notropis umbratilus*[c]									
Bluntnose minnow	*Pimephales notatus*	•						•	•	•
Fathead minnow	*Pimephales promelas*							•	•	•
Rudd	*Scardinius erythrophthalmus*							•	•	•
Fallfish	*Semotilus corporalis*									•
Creek chub	*Semotilus atromaculatus*							•		•
White sucker	*Catostomus commersoni*	•		•	•	•	•	•	•	•

Table 3-7. (cont.)

Common Name	Species	Year Captured 1927	1946	1969	1980	1989	1990	1991	1993	1994
White sucker	*Catostomus commersoni*	•		•	•	•	•	•	•	•
Northern hog sucker	*Hypentelium nigricans*									•
Redhorse	*Moxostoma sp.*	•	•	•	•	•	•	•	•	•
Yellow bullhead	*Ameiurus natalis*						•	•	•	•
Brown bullhead	*Ameiurus nebulosus*			•	•	•	•	•	•	•
Channel catfish	*Ictalurus punctatus*		•	•	•	•	•	•	•	•
American eel	*Anguilla rostrata*							•	•	
Banded killifish	*Fundulus diaphanus*	•	•			•	•	•	•	•
Burbot	*Lota lota*						•			
Brook silverside	*Labidesthes sicculus*					•	•	•		•
Brook stickleback	*Culaea inconstans*			•			•	•		
White perch	*Morone americana*			•	•	•	•	•	•	•
White bass	*Morone chrysops*		•			•	•	•	•	
Rock bass	*Ambloplites rupestris*					•	•	•	•	•
Green sunfish	*Lepomis cyanellus*							•		
Pumpkinseed	*Lepomis gibbosus*	•		•	•	•	•	•	•	•
Bluegill	*Lepomis macrochirus*			•	•	•	•	•	•	•
Smallmouth bass	*Micropterus dolomieui*			•	•	•	•	•	•	•
Largemouth bass	*Micropterus salmoides*	•			•	•	•	•	•	•
White crappie	*Pomoxis annularis*				•	•	•	•		•
Black crappie	*Pomoxis nigromaculatus*				•	•	•	•	•	•
Yellow perch	*Perca flavescens*	•	•	•	•	•	•	•	•	•
Walleye	*Stizostedion vitreum*		•	•	•	•	•	•	•	•
Tesselated darter	*Etheostoma nigrum*							•	•	•
Logperch	*Percina caprodes*		•				•	•	•	•
Freshwater drum	*Aplodinotus grunniens*			•	•	•	•	•	•	•

Source: Tango and Ringler (1996)

Notes: [a] Species captured using different methods as described in Tango and Ringler (1996).

[b] Splake is a hybrid of brook trout (*Salvelinus fontinalis*) and lake trout (*Salvelinus namaycush*).

[c] Species reported as captured by PTI (1993c), time of capture unknown.

Table 3-8. Levels of Natural Fish Reproduction in Onondaga Lake in 1991, Based on Catches in Shoreline Seine Hauls

High success (> 1,000 juveniles)		
White perch	Bluegill	Golden shiner
Gizzard shad	Brook silverside	
Banded killifish	Pumpkinseed	
Moderate success (100 - 1,000 fish)		
Largemouth bass	Yellow perch	
Carp	Emerald shiner	
Low success (1- 100 fish)		
Smallmouth bass	Northern pike	Brown bullhead
Black crappie	Spotfin shiner	
No success or unknown (0 fish)		
Bowfin	Alewife	Redhorse shiner
Rudd	Rainbow trout	Bluntnose minnow
Brook stickleback	Chain pickerel	Central mudminnow
Fathead minnow	White crappie	Common shiner
Spottail shiner	Rock bass	Tesselated darter
Channel catfish	Redfin shiner	White bass
Brown trout	Creek chub	Longnose gar
Burbot	Logperch	Rainbow smelt
Green sunfish	Walleye	Freshwater drum
Anadromous/Catadromous spawners		
White sucker	Sea lamprey	American eel
Hybrid (non-reproductive)		
Splake	Tiger muskellunge	

Source: Auer et al. (1996a)

Table 3-9. Species of Amphibians and Reptiles Expected to be Found in Covertypes Surrounding Onondaga Lake

Common Name	Scientific Name	Habitat
Amphibians - Frogs		
American toad	*Bufo americanus*	T/W
Gray treefrog	*Hyla chrysoscelis/versicolor*	T/W
Spring peeper	*Pseudacris crucifer*	T/W
Bullfrog	*Rana catesbiana*	W/A
Green frog	*Rana clamitans*	W/A
Wood frog	*Rana sylvatica*	T/W
Northern leopard frog	*Rana pipiens*	T/W/U
Pickerel frog	*Rana plaustris*	W
Salamanders		
Spotted salamander	*Ambystoma maculatum*	T/W
Jefferson complex [a]	*Ambystoma jeffersoni x laterale*	T/W
Red-spotted newt	*Notophthalmus viridescens*	T/W/A
Northern dusky	*Desmognathus fuscus*	T/A
Alleghany dusky	*Desmognathus ochrophaeus*	T/A
Northern redback	*Plethodon cinereus*	T
Northern slimy	*Plethodon glutinosus*	T
Northern spring	*Gyrinophilusporphyriticus*	A
Two-lined	*Eurycea bislineata*	T/A
Reptiles - Snakes		
Northern water snake	*Nerodia sipedon*	W/A
Northern brown snake	*Storeria dekayi*	T/U
Northern redbelly snake	*Storeria occipitomaculata*	T
Eastern garter snake	*Thamnophis sirtalis*	T/W/U
Northern ringneck snake	*Diadophis punctatus*	T
Black rat snake	*Elaphe obsoleta*	T
Eastern milk snake	*Lampropeltis triangulum*	T/U
Turtles		
Common snapping turtle	*Chelydra serpentina*	W/A
Painted turtle	*Chrysemys picta*	W/A
Wood turtle [a]	*Clemmys insculpta*	T/W/A
Musk turtle	*Sternotherus odoratus*	W/A

Sources: Conant and Collins (1998); NYSDEC (2001b)

Note: [a] NYS species of special concern

Habitat: Each species is assigned the habitat codes where they are most likely to be found. Species can potentially be found in other habitats. See Appendix A for covertypes included in each habitat code.

Habitat codes: T = Terrestrial, W = Wetland, A = Aquatic, U = Urban

Table 3-10. Species of Amphibians and Reptiles Found Near Onondaga Lake Between 1994 and 1997

Common Name	Scientific Name	Life Stages Found
Amphibians		
American toad	*Bufo americanus*	Adults
Gray treefrog	*Hyla chrysoscelis/versicolor*	Adults
Spring peeper	*Pseudacris crucifer*	Juveniles, adults
Green frog	*Rana clamitans*	Larvae, juveniles, adults
Northern leopard frog	*Rana pipiens*	Larvae, juveniles, adults
Spotted salamander	*Ambystoma maculatum*	Larvae, adults
Red-spotted newt	*Notophthalmus viridescens*	Adults
Reptiles		
Northern water snake	*Nerodia sipedon*	Adults
Northern brown snake	*Storeria dekayi*	Neonates, adults
Eastern garter snake	*Thamnophis sirtalis*	Neonates, adults
Common snapping turtle	*Chelydra serpentina*	Eggs, adults
Painted turtle	*Chrysemys picta*	Eggs, adults
Musk turtle	*Sternotherus odoratus*	Adults

Source: Ducey et al. (1998); Ducey (1997); Ducey and Newman (1995)

Table 3-11. Bird Species Found in Covertypes Surrounding Onondaga Lake based on NYS Bird Breeding Atlas Data

Family	Common Name	Scientific Name	Breeding Status	Habitat
Ardeidae	Great blue heron	*Ardea herodias*	PO	W/A
	Green heron	*Butorides virescens*	C	W/A
Anatidae	American black duck	*Anas rubripes*	C	W/A
	Mallard	*Anas platyrhynchos*	C	W/A
	Wood duck	*Aix sponsa*	C	W/A
	Canada goose *	*Branta canadensis*	C	W/A
Cathartidae	Turkey vulture *	*Cathartes atratus*	PO	T
Accipitridae	Red-tailed hawk	*Buteo jamaicensis*	C	T/U
	Sharp-shinned hawk [a]	*Accipiter striatus*	PR	T
Falconidae	American kestrel	*Falco sparverius*	C	T/U
Tetraonidae	Ruffed grouse	*Bonasa umbellus*	PO	T
Meleagrididae	Wild turkey [1]	*Meleagris gallopavo*	C	T/U
Phasianidae	Ring-necked pheasant	*Phasianus colchicus*	C	T/W
Rallidae	Sora	*Porzana carolina*	C	W
	Virginia rail	*Rallus limicola*	C	W
Charadriidae	Killdeer	*Charadrius vociferus*	C	T/U
Scolopacidae	Spotted sandpiper	*Actitis macularia*	C	W
	American woodcock	*Scolopax minor*	PR	T
Columbidae	Mourning dove	*Zenaida macroura*	C	T/U
	Rock dove	*Columba livia*	C	T/U
Cuculidae	Black-billed cuckoo *	*Coccyzus erthropthalmus*	PR	T
Strigidae	Great horned owl	*Bubo virginianus*	C	T
Caprimulgidae	Common nighthawk [a]	*Chordeiles minor*	PR	T/U
Apodidae	Chimney swift	*Chaetura pelagica*	PR	T/U
Trochilidae	Ruby-throated hummingbird	*Archilochus colubris*	PO	T
Alcedinidae	Belted kingfisher	*Ceryle alcyon*	C	W
Picidae	Red-headed woodpecker [a]	*Melanerpes erythrocephalus*	PO	T
	Red-bellied woodpecker	*Melanerpes carolinus*	PO	T
	Downy woodpecker	*Picoides pubescens*	C	T/U
	Hairy woodpecker	*Picoides villosus*	C	T/U
Tyrannidae	Eastern wood-pewee	*Contopus virens*	PR	T/U
	Common flicker	*Colaptes auratus*	C	T/U
	Pileated woodpecker *	*Dryocopus pileatus*	PO	T/W
	Alder flycatcher	*Empidonax alnorum*	PR	T

Table 3-11. (cont.)

Family	Common Name	Scientific Name	Breeding Status	Habitat
	Willow flycatcher	*Empidonax traillii*	C	T
	Least flycatcher	*Empidonax minimus*	PR	T
	Eastern phoebe	*Sayornis phoebe*	PR	T/U
	Great crested flycatcher	*Myiarchus crinitus*	PR	T
	Eastern kingbird	*Tyrannus tyrannus*	C	T/W
Alaudidae	Horned lark [a]	*Eremophila alpestris*	C	T/U
Hirundinidae	Purple martin	*Progne subis*	PO	W
	Tree swallow	*Tachycineta bicolor*	C	W
	Northern rough-winged swallow	*Stelgidopteryx serripennis*	C	W
	Bank swallow	*Riparia riparia*	C	T/W
	Barn swallow	*Hirundo rustica*	C	T/U
Corvidae	Blue jay	*Cyanocitta cristata*	C	T/U
	American crow	*Corvus brachyrhynchos*	C	T/U
	Fish crow *	*Corvus ossifragus*	PR	W/A
Paridae	Black-capped chickadee	*Poecile atricapillus*	C	T/U
Sittidae	White-breasted nuthatch	*Sitta carolinensis*	C	T
	Red-breasted nuthatch *	*Sitta canadensis*	PR	T/U
Certhiidae	Brown creeper	*Certhia americana*	PR	T/W
Troglodytidae	House wren	*Troglodytes aedon*	C	T/U
	Marsh wren	*Cistothorus palustris*	C	W
Mimidae	Gray catbird	*Dumetella*	C	T/U
	Northern mockingbird	*Mimus polyglottos*	C	T/U
	Brown thrasher	*Toxostoma rufum*	PR	T
Turdidae	Veery	*Catharus fuscescens*	PO	T/U
	Wood thrush	*Hylocichla mustelina*	C	T/U
	American robin	*Turdus migratorius*	C	T/U
Sylviidae	Blue-gray gnatcatcher	*Polioptila caerulea*	C	T/W
Bombycillidae	Cedar waxwing	*Bombycilla cedrorum*	C	T
Sturnidae	European starling	*Sturnus vulgaris*	C	T/U
Vireonidae	Yellow-throated vireo	*Vireo flavifrons*	PR	T/U
	Warbling vireo	*Vireo gilvus*	C	T/U
	Red-eyed vireo	*Vireo olivaceus*	C	T/U
Parulidae	Yellow warbler	*Dendroica petechia*	C	T/U
	American redstart	*Setophaga ruticilla*	C	T
	Mourning warbler *	*Oporornis agilis*	PO	T

Table 3-11. (cont.)

Family	Common Name	Scientific Name	Breeding Status	Habitat
	Common yellowthroat	*Geothlypis trichas*	C	T/W
Parylidae	House sparrow	*Passer domesticus*	C	U
Ploceidae	Baltimore oriole	*Icterus galbula*	C	T/U
Icteridae	Bobolink	*Dolichonyx oryzivorus*	PR	T
	Red-winged blackbird	*Agelaius phoeniceus*	C	T/W
	Eastern meadowlark	*Sturnella magna*	C	T
	Common grackle	*Quiscalus quiscula*	C	T/U
	Brown-headed cowbird	*Molothrus ater*	C	T/U
Thraupidae	Scarlet tanager	*Piranga olivacea*	PR	T
Fringillidae	Northern cardinal	*Cardinalis cardinalis*	C	T/U
	Rose-breasted grosbeak	*Pheucticus ludovicianus*	C	T/U
	Indigo bunting	*Passerina cyanea*	PR	T
	Rufous-sided towhee	*Pipilo erythrophthalmus*	C	T
	Chipping sparrow	*Spizella passerina*	C	T/U
	Field sparrow	*Spizella pusilla*	C	T
	Savannah sparrow	*Passerculus sandwichensis*	C	T
	Song sparrow	*Melospiza melodia*	C	T/U
	Swamp sparrow	*Melospiza georgiana*	C	W
	Purple finch	*Carpodacus purpureus*	PR	T
	House finch	*Carpodacus mexicanus*	C	U
	American goldfinch	*Carduelis tristis*	C	T

Sources: Andrle and Carroll (1988); * NYS Breeding Bird Atlas Interim Data (NYSDEC, 2001a); [1] Stiles (2001)

Breeding Status: Breeding status categories are defined as in the Breeding Bird Atlas for NYS:

C = Confirmed breeding, PR = Probable breeding, PO = Possible breeding

Note: [a] NYS species of special concern

Habitat : Each species is assigned the habitat codes where they are most likely to be found. Species can potentially be found in other habitats. See Appendix A for covertypes included in each habitat code.

Habitat codes: T = Terrestrial, W = Wetland, A = Aquatic, U = Urban

Table 3-12. Additional Species of Birds Observed on Onondaga Lake and its Shoreline During the Summer of 1993, Not Listed in Table 3-11

Family (Subfamily)	Common Name	Scientific Name
Gaviidae	Common loon[a]	*Gavia immer*
Phalacrocoracidae	Double-crested cormorant	*Phalacrocorax auritus*
Anatidae		
(Anatinae)	Gadwall	*Anas strepera*
	Blue-winged teal	*Anas discors*
	American wigeon	*Anas americana*
	Northern shoveler	*Anas clypeata*
	Wood duck	*Aix sponsa*
(Anserinae)	Brant	*Branta bernicla*
(Aythyinae)	Greater scaup	*Aythya marila*
	Lesser scaup	*Aythya affinis*
(Cygninae)	Mute swan	*Cygnus olor*
(Merginae)	Common merganser	*Mergus merganser*
Pandionidae	Osprey[a]	*Pandion haliaetus*
Charadriidae	Semipalmated plover	*Charadrius semipalmatus*
Scolopacidae	Greater yellowlegs	*Tringa melanoleuca*
	Ruddy turnstone	*Arenaria interpres*
	Semipalmated sandpiper	*Calidris pusillus*
Laridae		
(Larinae)	Great black-backed gull	*Larus marinus*
	Ring-billed gull	*Larus delawarensis*
(Sterninae)	Common tern[b]	*Sterna hirundo*
	Caspian tern	*Sterna caspia*
Paridae	Tufted titmouse	*Parus bicolor*

Source: Tango (1993)

Notes: [a] New York State species of special concern.

[b] New York State threatened species.

Table 3-13. Species of Waterfowl Observed Wintering on Onondaga Lake from 1990 to 1999

Common Name	Scientific Name	Recorded Observations									
		1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Horned grebe	*Podiceps auritus*	•	•		•			•		•	•
Mallard	*Anas platyrhynchos*	•	•	•	•	•	•	•	•	•	•
Black duck	*Anas rubripes*	•	•	•	•	•	•	•	•	•	•
Gadwall	*Anas strepera*	•	•	•	•	•	•	•	•	•	•
Green-winged teal	*Anas crecca*	•	•	•	•	•	•	•	•	•	
Ring-necked duck	*Aythya collaris*		•		•						•
Greater scaup	*Aythya marila*		•		•	•		•	•		•
Lesser scaup	*Aythya affinis*			•	•		•				
Common goldeneye	*Bucephala clangula*	•	•	•	•	•	•	•	•	•	•
Common merganser	*Mergus merganser*	•	•	•	•	•	•	•	•	•	•
Red-breasted merganser	*Mergus serrator*	•	•	•	•	•			•	•	•
Great blue heron	*Ardea herodias*	•	•	•	•	•	•	•	•	•	•
Belted kingfisher	*Megaceryle alcyon*	•	•	•	•	•	•	•	•	•	•
Red-tailed hawk	*Buteo jamaicensis*	•	•	•	•	•	•	•	•	•	•
Osprey	*Pandion haliaetus*										
American coot	*Fulica americana*		•	•		•	•		•		•
Mute swan	*Cygnus olor*				•						

Sources: Onondaga Audubon Society (1990, 1991, 1992, 1993); Rusk (1994)
National Audubon Society; http://birdsource.tc.cornell.edu/cbcdata/ (November 20, 2001)

Table 3-14. Species of Mammals Expected to be Found in Covertypes Surrounding Onondaga Lake

Family	Common Name	Scientific Name	Habitat
Didelphidae	Virginia opossum	*Didelphis virginiana*	T/U
Soricidae	Shorttail shrew	*Blarina brevicauda*	T/U
	Masked shrew	*Sorex cinereus*	T/W/U
	Smoky shrew	*Sorex fumeus*	T/W
	Water shrew	*Sorex palustris*	W
Talpidae	Hairy-tailed mole	*Parascalops breweri*	T
	Star-nosed mole	*Condylura cristata*	W
Vespertilionidae	Little brown bat	*Myotis lucifugus*	T
	Small-footed bat [a]	*Myotis leibii*	T
	Northern long-eared bat	*Myotis septentrionalis*	T
	Indiana bat [b]	*Myotis sodalis*	T
	Big brown bat	*Eptesicus fuscus*	T
	Red bat	*Lasiurus borealis*	T
	Hoary bat	*Lasiurus cinereus*	T
	Silver-haired bat	*Lasionycteris noctivagans*	T
	Eastern pipistrelle	*Pipistrellus subflavus*	T
Leporidae	Eastern cottontail	*Sylvilagus floridanus*	T/U
Sciuridae	Eastern chipmunk	*Tamias striatus*	T
	Woodchuck	*Marmota monax*	T/U
	Gray squirrel	*Sciurus carolinensis*	T/U
	Southern flying squirrel	*Glaucomys volans*	T
	Northern flying squirrel	*Glaucomys sabrinus*	T
	Red squirrel	*Tamiasciurus hudsonicus*	T
Castoridae	Beaver	*Castor canadensis*	W
Muridae	Norway rat	*Rattus norvegicus*	U
	White-footed mouse	*Peromyscus leucopus*	T/U
	Deer mouse	*Peromyscus maniculatus*	T
	Red-backed vole	*Clethrionomys gapperi*	T/W
	Meadow vole	*Microtus pennsylvanicus*	T/W
	Woodland vole	*Microtus pinetorum*	T
	House mouse	*Mus musculus*	U

Table 3-14. (cont.)

Family	Common Name	Scientific Name	Habitat
	Muskrat	*Ondatra zibethicus*	A
	Southern bog lemming	*Synaptomys cooperi*	T/W
Dipodidae	Woodland jumping mouse	*Napaeozapus insignis*	T
	Meadow jumping mouse	*Zapus hudsonius*	T/W
Canidae	Coyote	*Canis latrans*	T
	Red fox	*Vulpes fulva*	T
	Gray fox	*Urocyon cinereoargenteus*	T
Procyonidae	Raccoon	*Procyon lotor*	T/U/W
Mustelidae	Mink	*Mustela vison*	W/A/T
	Ermine	*Mustela ermina*	T
	Long-tailed weasel	*Mustela frenata*	T
	River otter	*Lutra canadensis*	W/A
	Striped skunk	*Mephitis mephitis*	T/U
Cervidae	White-tailed deer	*Odocoileus virginianus*	T/U/W

Source: Kurta (1995)

Notes: [a] NYS species of special concern, [b] NYS endangered species

Habitat: Each species is assigned the habitat codes where they are most likely to be found. Species can potentially be found in other habitats. See Appendix A for covertypes included in each habitat code.

Habitat codes: T = Terrestrial, W = Wetland, A = Aquatic, U = Urban

4. SCREENING-LEVEL PROBLEM FORMULATION AND ECOLOGICAL EFFECTS EVALUATION (ERAGS STEP 1)

This initial ecological screening assessment includes a screening-level problem formulation and an ecological-effects evaluation (USEPA, 1997a), which are presented in this chapter. These components are then used to complete the screening-level exposure estimate and risk calculations (ERAGS Step 2) contained in Chapter 5.

The site description, required for Step I of the FWIA (NYSDEC, 1994a) and used to assist in this screening-level problem formulation, was included in Chapter 3. A summary of chemical contamination at the site and around the lake, which is a component of ERAGS Step 1, has been included in Chapter 2 and in the remedial investigation (RI) report (TAMS, 2002b).

Honeywell largely completed the initial screening-level problem formulation for Onondaga Lake during preparation of the Onondaga Lake RI/FS Work Plan (PTI, 1991), based on a review of existing information for the lake. As part of the work plan, Honeywell developed a conceptual site model, identified preliminary chemicals of potential concern/stressors of potential concern (COPCs/SOPCs) and representative ecological receptors, defined assessment and measurement endpoints, formulated the objectives of the BERA, and developed a study design to collect the data needed to satisfy the BERA objectives.

Several elements of the screening-level problem formulation have been refined by Honeywell and NYSDEC since the work plan was completed in 1991, based on information collected during the 1992 and 1999/2000 Honeywell RI field investigations and more recent investigations, such as the 2002 sampling conducted by NYSDEC.

In developing the contents of this BERA, several exchanges have occurred between Honeywell (formerly AlliedSignal) and NYSDEC since the RI/FS Work Plan was finalized in 1991 (e.g., PTI, 1995a,b; Larson, pers. comm., 1995, pers. comm., 1996). The relevant content of these exchanges, NYSDEC comments (submitted in March 1999) on the May 1998 draft BERA, and the results of the subsequent meetings have been incorporated into this document.

The following sections present the major components of the initial problem formulation, including:

- Development of a preliminary conceptual site model, including contaminant fate and transport and complete exposure pathways.
- Preliminary identification of COPCs/SOPCs.
- Preliminary identification of representative ecological receptors.
- Preliminary identification of assessment and measurement endpoints.

- Preliminary ecological-effects evaluation and the establishment of conservative contaminant exposure levels.

4.1 Preliminary Conceptual Site Model

The preliminary conceptual site model for the Onondaga Lake BERA, presented in Figure 4-1, is the final version of the conceptual model presented in the Onondaga Lake RI/FS Work Plan (PTI, 1991). The preliminary conceptual site model identifies the following:

- Primary and secondary sources.
- Potential pathways.
- Major chemical/stressor groups.
- Potential exposure routes and receptors.
- Effects to be initially evaluated as part of the BERA.

As described in Chapter 1, Comprehensive Environmental Response Compensation and Liability Act of 1980- (CERCLA-) related stressors are referred to as chemicals, whereas non-CERCLA stressors, such as chloride, phosphorus, depleted dissolved oxygen (DO), and reduced water transparency, are referred to as stressors. The term "contaminants" is also used throughout this document to describe these substances, and chemical contaminants in particular.

Through the primary conceptual model, Honeywell identified that primary sources of contaminants and stressors to Onondaga Lake are point-source discharges, including tributaries, and non-point sources, including groundwater. Although the atmosphere may be an additional source of some substances, atmospheric inputs into the lake are considered minor as compared to point-source and other non-point sources discharges. Significant point-source discharges to the lake, including tributaries, are the Honeywell sources (e.g., the East Flume and Interstate 690 [I-690] outfalls) and the Metropolitan Syracuse Sewage Treatment Plant (Metro). The larger tributaries to the lake are Onondaga Creek, Ninemile Creek, Ley Creek, and Harbor Brook. Smaller tributaries include Bloody Brook, Sawmill Creek, and Tributary 5A. Honeywell facilities and disposal areas near Onondaga Lake are described in Chapter 2 of this report and in the RI (TAMS, 2002b).

After chemical contaminants enter Onondaga Lake, they are distributed among the water, sediments, floodplain soils (including wetlands), and biota. Contaminants enter the sediment by deposition or precipitation from the water column. Deposition is usually facilitated by adsorption to particles or incorporation into planktonic organisms that eventually die and sink to the bottom of the lake. Precipitation of substances is controlled primarily by the temperature and chemical composition of the lake water. Contaminants are deposited onto adjacent wetlands and floodplain soils from lake tributaries during high flows or via hydrologic connections with the lake.

Water, sediment, soil, and biota may then become secondary sources of contamination by releasing compounds to aquatic, terrestrial, and human receptors (Figure 4-1). Receptors may be exposed to contaminants by absorption from the water column through dermal layers or respiratory organs and ingestion via food, sediment, soil, or water.

The stressors in Onondaga Lake include nutrients (i.e., nitrite, phosphorous, sulfide), calcite, chloride, salinity, ammonia, depleted DO, reduced transparency, and wave scour. Calcium, chloride, and sodium are associated with ionic waste inputs into the lake from former Honeywell facilities, as well as natural sources. Many of the lake nutrients originate from sewage that is discharged from the Metro outfalls or the combined sewer overflows (CSOs) that discharge into lake tributaries (e.g., Onondaga Creek, Ley Creek, Bloody Brook, and Harbor Brook). Within the lake, secondary sources of stressors include water and sediment. The extremely high concentrations of calcite in the lake are due to soda-ash manufacturing activities (see the RI for details [TAMS, 2002b]).

Stressors, such as salinity, reduced transparency, and depleted DO, are associated with the pollution of Onondaga Lake. Wave-scour stress can be associated with lake-level management, although over an approximately ten-year period from 1983 to 1992 the lake level has been fairly consistent, with a difference between minimum lake elevations of 0.6 ft (18 cm) and a difference between maximum lake elevations of 3.2 ft (98 cm) (Table 3-1). The Phoenix Dam regulates the water level of Onondaga Lake.

4.1.1 Preliminary Identification of Chemicals/Stressors of Potential Concern

Preliminary COPCs/SOPCs are divided into two categories: 1) those identified by Honeywell in the RI/FS Work Plan that was finalized in 1992, and 2) those based on results of data collected by Honeywell during the 1992, 1999, and 2000 RI field investigations, or on results of more recent investigations, such as the 2002 wetland sampling, conducted by NYSDEC (Table 4-1). As described earlier, the COPCs/SOPCs include both CERCLA-related and non-CERCLA-related chemicals and stressors.

4.1.1.1 Chemicals of Potential Concern

The chemical contaminant that has historically received the most attention in Onondaga Lake is mercury, which was used in Honeywell's chlor-alkali process. However, numerous other potentially toxic chemicals, including cadmium; chromium; copper; lead; nickel; zinc; polychlorinated biphenyls (PCBs); polycyclic aromatic hydrocarbon (PAH) compounds; benzene, toluene, ethylbenzene, and xylenes (BTEX); chlorinated benzenes; and dioxins/furans have been found at elevated concentrations in various lake media. A preliminary list of chemicals of potential concern is provided in Table 4-1, with the COPCs identified in the original work plan listed separately. The screening-level exposure estimates consider all contaminants detected during sampling, which is a larger group of compounds than identified in this preliminary step (see Chapter 5). Chemicals with the potential to bioaccumulate or biomagnify in the food chain are of particular concern in the ecological risk assessment.

4.1.1.2 Stressors of Potential Concern

The stressors in Onondaga Lake include nutrients (i.e., nitrite, phosphorus, sulfide), calcium, chloride, salinity, ammonia, depleted DO, reduced transparency, and oncolites (Table 4-1). Of these, depleted DO, nitrogen, phosphorus, and sulfide were added to the initial work plan SOPC list after potential problems related to those eutrophication-related variables were identified (Effler et al., 1996a). Salinity was added after concerns were expressed that this variable may have affected various kinds of biological communities in the lake (Auer et al. 1996a). Oncolites were added after they were identified as a potential limiting factor to macrophytes in shallow parts of Onondaga Lake (Auer et al., 1996a).

4.1.1.3 Ionic Waste Discharges

A class of substances that has been historically discharged to Onondaga Lake is the ionic waste that was produced as a result of Honeywell's soda-ash manufacturing process and pumped to the Honeywell wastebeds in the form of a slurry (5 to 10 percent suspended solids). Ionic waste overflow from some, if not all, of the Honeywell wastebeds has drained off and entered Onondaga Lake over the last 100 years (PTI, 1991). The overflow, contaminated with calcium, chloride, and sodium ions entered the lake, primarily via Ninemile Creek (Effler and Harnett, 1996). Solvay waste was also discharged into the lake (e.g., via the East Flume; see RI Chapter 4, Section 4.5.1 [TAMS, 2002b]), with the solids forming a substantial delta in the area of the lake in front of Wastebed B. See Chapter 4 of the Onondaga Lake RI report for additional information on the Solvay Wastebeds and the Honeywell in-lake waste disposal.

Although the amount of ionic waste entering the lake has decreased since the 1987 closure of the Honeywell facility, large quantities of ionic waste remain in and continue to be released to the lake. The various components of this waste and the potential risks they pose to ecological receptors in and around the lake are evaluated in this BERA. For evaluation purposes, ionic waste is considered as part of the total input of individual ions (e.g., calcium, chloride), rather than as components of a separate class of substances termed "ionic waste." The potential risks of ionic waste are evaluated in the BERA as follows:

- All ions: these chemicals were evaluated as a group in the BERA as components of the salinity of lake water, which undermines water quality. These chemicals were also evaluated as a group in the RI as potential contributors to lake stratification.

- Chloride: this chemical was evaluated individually as a stressor in lake and tributary water because it has been found to be toxic at elevated concentrations to various groups of aquatic organisms.

- Calcium: this chemical was evaluated individually as a stressor in sediments, due to the contamination of lake sediments with calcium, as well as the formation of oncolites. Oncolites have formed in the lake as a result of the calcium-contaminated discharge of ionic waste during the production of soda ash (Dean and Eggleston, 1984). Oncolite formation is likely to adversely affect fish spawning success and/or impede the establishment of macrophyte communities. Calcite precipitates alter aquatic habitats in Onondaga Lake by reducing transparency in the lake, which causes reductions in photosynthesis.

4.1.2 Preliminary Identification of Ecological Receptors

The key groups of ecological receptors considered in the BERA include representatives of major trophic groups that are found in and around Onondaga Lake. These groups, which were identified in the Onondaga Lake RI/FS Work Plan (PTI, 1991) and refined in later documents and through discussions with NYSDEC, include:

- Aquatic macrophytes.
- Phytoplankton.
- Zooplankton.
- Terrestrial plants.
- Benthic macroinvertebrates.
- Amphibians and reptiles.
- Fish.
- Insectivorous birds, such as the tree swallow (*Tachycineta bicolor*).
- Benthivorous birds, such as the mallard (*Anas platyrhynchos*).
- Piscivorous birds, such as the belted kingfisher (*Ceryle alcyon*), great blue heron (*Ardea herodias*), and osprey (*Pandion haliaetus*).
- Carnivorous birds, such as the red-tailed hawk (*Buteo jamaicensis*).
- Insectivorous mammals, such as the little brown bat (*Myotis lucifugus*) and short-tailed shrew (*Blarina brevicauda*).
- Piscivorous mammals, such as the mink (*Mustela vison*) and river otter (*Lutra canadensis*).

Groups that are not covered by these receptors, such as herbivorous birds and mammals and omnivorous birds and mammals, are considered to be at lower risk than some of the receptors selected, based on their feeding habits. Generally, concentrations of bioaccumulative contaminants are lower in plants and the animals feeding on them than in higher-level trophic organisms. Therefore, use of the receptors identified above is considered to be protective of most of the flora and fauna found in and around Onondaga Lake.

4.1.3 Preliminary Identification of Assessment and Measurement Endpoints

The preliminary assessment and measurement endpoints evaluated in this BERA are presented in Table 4-2.

4.1.3.1 Assessment Endpoints

Assessment endpoints are explicit expressions of the actual environmental values that are to be protected, operationally defined by an ecological entity and its attributes (USEPA, 1998). They are expressed in terms of the ecological receptor (e.g., local population of a particular species, community of organisms, or other ecosystem component) and an attribute (e.g., survival or reproduction). Communities and populations selected for the endpoints represent receptors in the absence of COPC and SOPC inputs. Assessment endpoints include:

- Sustainability of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife.
- Sustainability of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife.
- Sustainability of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife.
- Sustainability of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife.
- Sustainability of a benthic invertebrate community that can serve as a food source for local fish and wildlife.
- Sustainability (i.e., survival, growth, and reproduction) of local fish populations.
- Sustainability (i.e., survival, growth, and reproduction) of local amphibian and reptile populations.
- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local benthivorous waterfowl populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local carnivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous mammalian populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous mammalian populations.

Final assessment endpoints are selected in Step 3 of ERAGS, contained in Chapter 6 of this report.

4.1.3.2 Measurement Endpoints

Measurement endpoints are the measurable changes in an attribute of an assessment endpoint or in response to a chemical/stressor to which a receptor is exposed. Measurement endpoints include expressions such as toxicity test results, benthic community diversity measures, contaminant concentration in exposure media, and field observations. It is common practice to use more than one measurement endpoint to evaluate each assessment endpoint, when possible.

Specific measurement endpoints associated with each assessment endpoint are established in Step 3 of the ERAGS process, which is contained in Chapter 6 of this report. General measurement endpoints to be considered in this risk assessment relative to assessment endpoints are:

- Field observations of community structure and abundance (aquatic macrophyte, phytoplankton, zooplankton, benthic invertebrate, fish, amphibian, and reptile) in relation to measured concentrations of contaminants and stressors.

- Measured concentrations of COPCs/SOPCs in surface water as compared to NYSDEC, USEPA, and other water quality standards, criteria, and guidance for aquatic life (see Chapter 3, Section 3.4).

- Measured concentrations of COPCs/SOPCs in sediment as compared to NYSDEC, USEPA, site-specific, and other sediment-quality guidelines for aquatic life (see Chapter 3, Section 3.4).

- Measured concentrations of COPCs in soil as compared to USEPA and/or other guidance (see Chapter 3, Section 3.4).

- Laboratory (greenhouse studies) and field experiments measuring macrophyte growth and survival.

- Sediment toxicity to aquatic invertebrates based on laboratory tests of field-collected sediments using standard laboratory test species and protocol for survival, growth, and reproductive endpoints.

- Benthic invertebrate community indices, such as richness, abundance, diversity, and biomass.

- Measured fish tissue concentrations as compared to toxicity values found in peer-reviewed literature.

- Observed effects on fish foraging and nesting.

- Field observations of deformation or disease in fish.

- Modeled dietary doses of COPCs, based on measured concentrations of COPCs in lake media (surface water, sediment, and prey), as compared to toxicity reference values (TRVs) for aquatic food-chain receptors.

- Modeled dietary doses of COPCs, based on measured concentrations of COPCs in lake-related media (surface water, soils, and prey), as compared to toxicity reference values for terrestrial food-chain receptors.

4.2 Screening-Level Ecological-Effects Evaluation

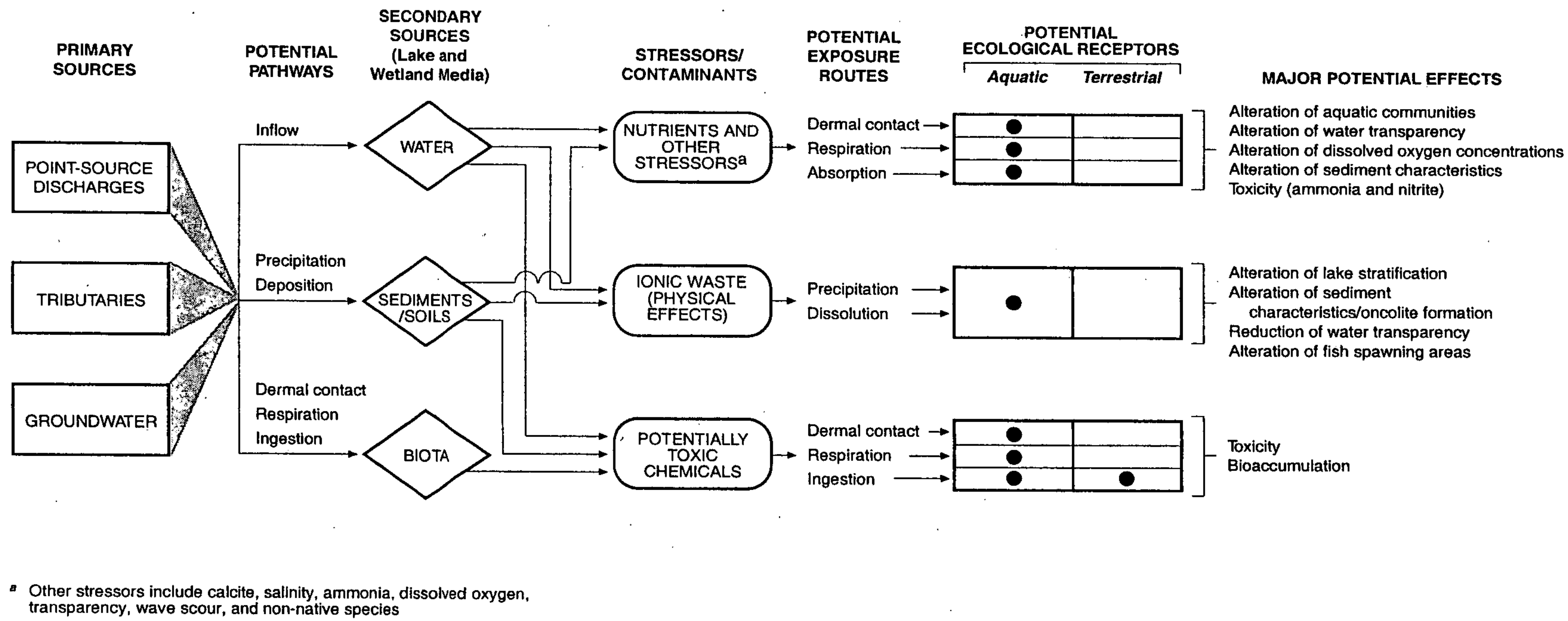
The screening-level ecological-effects evaluation establishes contaminant exposure levels that represent conservative thresholds for adverse ecological effects. For each complete exposure pathway, route, and contaminant, a screening ecotoxicity value is selected. Details of the ecological screening are provided in Appendix D. NYSDEC and USEPA values were the primary screening values used for surface water (Tables 4-3 [organics] and 4-4 [inorganics]), sediments (Tables 4-5 [dry weight] and 4-6 [organic carbon-normalized]), and soils (Table 4-7). These values were supplemented with values from the Ontario Ministry of the Environment (Persaud et al., 1993) and the Oak Ridge National Laboratory (ORNL) (Jones et al., 1997) for some media. Soil benchmarks developed by ORNL (Efroymson et al., 1997a) were used to screen plants (Table 4-8).

Toxicity values for fish tissue were not readily available; therefore, measures of toxicity in fish tissue from NYSDEC (Newell et al., 1987), the International Joint Commission (IJC) of the United States and Canada (IJC, 1988), and ORNL (Sample et al., 1996) were used for screening (Table 4-9).

For wildlife receptors a screening ecotoxicity value was selected for each complete exposure pathway, route, and contaminant. Consistent with USEPA guidance (1997a), no observed adverse effect level (NOAEL) toxicity values were used for avian and mammalian receptors, when available, to ensure that risk was not underestimated. When only lowest observed adverse effect level (LOAEL) toxicity values were available, a correction factor of 0.1 was applied. Table 4-10 contains toxicity values used to screen avian receptors and Table 4-11 contains values used for mammalian screening. The primary literature sources used to select toxicity values include Sample et al. (1996), Newell et al. (1987), and values presented in Honeywell's revised draft BERA (Exponent, 2001b).

For wildlife toxicity values, the most conservative value available for each class (e.g., avian, mammal) was used. When toxicity values were only available for one wildlife class (i.e., mammals or birds), those values were used for both classes for screening purposes only. If a toxicity value was not available for a compound, toxicity values for compounds with similar physical/chemical characteristics were used.

Several of the COPCs did not have any published toxicity values available, and alternate toxicity values were considered inappropriate. Therefore these compounds were not carried through to the final quantitative assessment performed for the risk characterization (Chapter 10), but are discussed in Chapter 11, Uncertainty Analysis.



Source: Modified from Exponent, 2001b

Figure 4-1. Conceptual Site Model for the Baseline Ecological Risk Assessment for Onondaga Lake

Table 4-1. Preliminary List of Chemicals and Stressors of Potential Concern for the Onondaga Lake BERA

	Environmental Medium[a,b]						
	Water		Sediment		Soil	Tissue	
	Work Plan	New	Work Plan	New	New	Work Plan	New
COPCs							
Metals							
Aluminum							•
Antimony				•	•		•
Arsenic					•		•
Barium		•			•		•
Berylium					•		•
Cadmium	•		•		•		•
Chromium	•		•		•		•
Copper	•		•		•		
Iron					•		
Lead	•		•		•		•
Manganese		•			•		
Mercury/Methylmercury	•		•		•	•	
Nickel	•		•				
Selenium					•		•
Silver				•			•
Thallium					•		•
Vanadium					•		•
Zinc	•		•				•
Cyanide	•						
Volatile and Semivolatile Organic Compounds							
Benzene	•		•				
Toluene		•		•			
Xylenes		•		•			
Chlorobenzene	•		•		•		
Dichlorobenzenes	•		•		•		•
Trichlorobenzenes	•		•		•		•
Ethylbenzene				•			
PAH Compounds (total)				•	•		•
Bis(2-ethylhexyl)phthalate		•					
Hexachlorobenzene				•	•		•
2-Methylnaphthalene				•			
Phenol				•	•		

Table 4-1. (cont.)

	Environmental Medium[a,b]						
	Water		Sediment		Soil	Tissue	
	Work Plan	New	Work Plan	New	New	Work Plan	New
Pesticides/Polychlorinated Biphenyls							
Total PCBs			•		•	•	
DDT and metabolites				•	•		•
Aldrin				•			•
Chlordane (total)				•	•		
alpha-endosulfan				•			
Endrin				•			•
Heptachlor/heptachlor epoxide				•	•		•
Hexachlorocyclohexanes (a,b,y, and sum)				•	•		
Dioxins/furans				•	•		•
SOPCs							
Calcium[c]			•				
Oncolites[c]				•			
Chloride[c]	•						
Salinity[c]		•					
Ammonia	•		•				
Nitrite		•					
Phosphorus		•					
Sulfide		•		•			
Dissolved oxygen		•					
Transparency[c]	•						

Notes: • – identified as a COPC/SOPC.

1. COPC - Chemical of Potential Concern
2. PAH - Polycyclic Aromatic Hydrocarbon
3. PCB - Polychlorinated Biphenyl
4. SOPC - Stressor of Potential Concern

[a] Environmental medium in which chemical or stressor was evaluated.

[b] Chemicals and stressors are denoted as having been identified in the RI/FS work plan (PTI, 1991) or as having been added after the work plan was finalized (i.e., new).

[c] Stressors potentially related in part to ionic waste.

Table 4-2. Preliminary Assessment and Measurement Endpoints for the Onondaga Lake BERA

Receptors	Assessment Endpoints	Measurement Endpoints
Aquatic Macrophytes	Sustainability of community as source of food and shelter for local fauna	Field observations of community abundance and compostion Measured concentrations of COCs/SOCs in surface water as compared to criteria/guidelines Onondaga Lake macrophyte tranplant and laboratory studies
Phytoplankton	Sustainability of community as source of food for local fauna	Field observations of community compostion Measured concentrations of COCs/SOCs in surface water as compared to criteria/guidelines
Zooplankton	Sustainability of community as source of food for local fauna	Field observations of community compostion Measured concentrations of COCs/SOCs in surface water and sediments as compared to criteria/guidelines Onondaga Lake literature studies
Terrestrial Plants	Sustainability of community as source of food and shelter for local fauna	Field observations Measured concentrations of COCs/SOCs in sediments/soil as compared to criteria/guidelines
Benthic Macroinvertebrates	Sustainability of community as source of food for local fish and wildlife	Field observations of community abundance and compostion Sediment toxicity tests Measured concentrations of COCs/SOCs in surface water and sediment as compared to criteria/guidelines
Fishes	Protection/maintenance of local populations[a]	Field observations of community abundance and composition Measured COCs/SOCs in water, sediment, and tissue as compared to criteria, guidelines and TRVs Histopathology, frequency of disease and deformation
Amphibians/reptiles	Protection/maintenance of local populations[a]	Field observations of community abundance and compostion Measured concentrations of COCs/SOCs in surface water and sediments/soil as compared to criteria/guidelines Onondaga Lake literature studies

Table 4-2. (cont.)

Receptors	Assessment Endpoints	Measurement Endpoints
Birds[b]	Protection/maintenance of local populations[a]	Measured COCs/SOCs in water, sediment, soil, and fish tissue as compared to criteria/guidelines Field observations of community abundance and composition COC body burdens based on food web models as compared to TRVs
Mammals[c]	Protection/maintenance of local populations[a]	Measured COCs/SOCs in water, sediment, soil, and fish tissue as compared to criteria/guidelines Field observations of community abundance and composition COC body burdens based on food web models as compared to TRVs

Notes:

1. COC/SOC = chemical/stressor of concern
2. SQV = sediment quality value
3. TRV = toxicity reference value
4. WQV = water quality value
5. Communities and populations are assumed to be those in the absence of widespread contamination.

[a] Survival, growth, and reproduction.

[b] Includes benthivorous, piscivorous, insectivorous, and carnivorous birds.

[c] Includes piscivorous and insectivorous mammals.

Table 4-3. Ecological Screening Values Used for Organic Chemicals in Surface Water of Onondaga Lake[a]

Chemical	Conc. Units	USEPA AWQC-FCV (Aquatic Life)	USEPA Chronic (Aquatic Life)	USEPA Acute (Aquatic Life)	USEPA CCC (Aquatic Life)	USEPA CMC (Aquatic Life)	NYSDEC Acute (Aquatic)	NYSDEC Chronic (Aquatic)	USEPA Tier II (Aquatic Life)
Conventional Parameters									
Total chloride	mg/L				230	860			
Volatile Organic Compounds									
Aromatic Hydrocarbons									
Benzene	μg/L								46
Toluene	μg/L								130
Ethylbenzene	μg/L								290
Xylene isomers (total)	μg/L								1.8
Chlorinated Aromatic Hydrocarbons									
Chlorobenzene	μg/L							5	130
1,2-Dichlorobenzene	μg/L								14
1,3-Dichlorobenzene	μg/L								71
1,4-Dichlorobenzene	μg/L								15
Dichlorobenzenes (sum)	μg/L		763	1,120				5	
1,2,4-Trichlorobenzene	μg/L								110
Trichlorobenzenes (sum)	μg/L							5	
Halogenated Alkanes									
1,1-Dichloroethane	μg/L								47
1,1,1-Trichloroethane	μg/L		9,400	18,000					62
1,2-Dichloropropane	μg/L		5,700	23,000					
1,1,2-Trichloroethane	μg/L		9,400	18,000					
1,1,2,2-Tetrachloroethane	μg/L		2,400	9,320					420
Halogenated Alkenes									
cis-1,3-Dichloropropene	μg/L		244	6,060					
trans-1,3-Dichloropropene	μg/L		244	6,060					
Trichloroethene	μg/L		21,900	45,000					350
Tetrachloroethene	μg/L		840	5,280					120

Table 4-3. (cont.)

Chemical	Concentration Units	USEPA AWQC-FCV (Aquatic Life)	USEPA Chronic (Aquatic Life)	USEPA Acute (Aquatic Life)	USEPA CCC (Aquatic Life)	USEPA CMC (Aquatic Life)	NYSDEC Acute (Aquatic)	NYSDEC Chronic (Aquatic)	USEPA Tier II (Aquatic Life)
Semivolatile Organic Compounds									
Chlorinated Aromatic Hydrocarbons									
Hexachlorobenzene	µg/L		3.68	6					
Low Molecular Weight Polycyclic Aromatic Hydrocarbons									
Naphthalene	µg/L		620	2,300					24
Acenaphthene	µg/L	23	520	1,700					
Fluorene	µg/L								3.9
Phenanthrene	µg/L	6.3							
High Molecular Weight Polycyclic Aromatic Hydrocarbons									
Fluoranthene	µg/L	8.1							
Benzo[a]pyrene	µg/L								0.014
Phenols									
Phenol	µg/L		2,560	10,200					
Substituted Phenols									
2-Chlorophenol	µg/L			4,380					
2,4-Dichlorophenol	µg/L		365	2,020					
2,4,6-Trichlorophenol	µg/L		970						
2,4,5-Trichlorophenol	µg/L		63	100					
Pentachlorophenol	µg/L	13			15	19	19.5	15	
2-Nitrophenol	µg/L		150	230					
4-Nitrophenol	µg/L		150	230					
Chlorinated Aliphatic Hydrocarbons									
Hexachloroethane	µg/L		540	980					12
Hexachlorobutadiene	µg/L		9.3	90				1	
Hexachlorocyclopentadiene	µg/L		5.2	7				0.45	
Halogenated Ethers									
4-Bromophenyl-phenyl ether	µg/L								1.5
Phthalates									
Diethyl phthalate	µg/L								220
Di-*n*-butyl phthalate	µg/L								33
Butylbenzyl phthalate	µg/L								19
bis[2-Ethylhexyl]phthalate	µg/L							0.6	32

Table 4-3. (cont.)

Chemical	Conc. Units	USEPA AWQC-FCV (Aquatic Life)	USEPA Chronic (Aquatic Life)	USEPA Acute (Aquatic Life)	USEPA CCC (Aquatic Life)	USEPA CMC (Aquatic Life)	NYSDEC Acute (Aquatic)	NYSDEC Chronic (Aquatic)	USEPA Tier II (Aquatic Life)
Miscellaneous Oxygenated Compounds									
Dibenzofuran	μg/L								20
Organonitrogen Compounds									
2,4-Dinitrotoluene	μg/L		230	330					
Pesticides/Polychlorinated Biphenyls									
γ-Hexachlorocyclohexane	μg/L	0.08	0.08			0.95	0.95		
Aldrin	μg/L					3			
α-Chlordane	μg/L				0.0043	2.4			
γ-Chlordane	μg/L				0.0043	2.4			
Dieldrin	μg/L	0.062			0.056	0.24			
α-Endosulfan	μg/L							0.009	0.051
β-Endosulfan	μg/L							0.009	0.051
Endosulfan (sum of α- and β-)	μg/L				0.056	0.22			
Endosulfan sulfate	μg/L								
Endrin	μg/L	0.061			0.036	0.086	0.086	0.036	
Heptachlor	μg/L				0.0038	0.52			0.0069
Heptachlor epoxide	μg/L				0.0038	0.52			
Methoxychlor	μg/L				0.03			0.03	0.019
4,4'-DDT	μg/L				0.001	1.1			0.013
Toxaphene	μg/L				0.0002	0.73		0.005	0.011
Polychlorinated biphenyls (sum)	μg/L			2	0.014			0.00012	0.19
Dioxins/Furans									
Total dioxins	μg/L		0.0056						

[a] Guidelines are applied to both the dissolved and unfiltered forms.

NYSDEC Water Quality Standards (NYSDEC 1999a)
Dichlorobenzenes standard applied to the sum of 1,2-, 1,3-, and 1,4-dichlorobenzene.
Trichlorobenzes standard applied to the sum of 1,2,3-, 1,2,4-, and 1,3,5-trichlorobenzene.
Pentachlorophenol standards calculated to reflect site-specific pH (average in 1998 = 7.8).
Standard for "sum of p,p'-DDT, p,p'-DDE, and p,p'-DDD" applied to DDT and metabolites (Sum).
Endosulfan standard applied separately to alpha-endosulfan and beta-endosulfan.
PCB standard is for wildlife protection (vs. specific acute or chronic effects) and applies to sum of these substances.

Table 4-3. (cont.)

USEPA Surface Water Benchmarks (USEPA, 1996a)
AWQC/FCV - EPA chronic ambient water quality criteria or EPA-derived final chronic values.
Tier II - Values calculated using Great Lakes Water Quality Initiative Tier II methodology (40 CFR 9, 122, 123, 131, and 132 [1995]).
All Tier II values as calculated in Suter and Mabrey (1994), except DDT and heptachlor (EPA support documents cited) and the following (calculated for USEPA [1996a]): 4-bromophenyl phenyl ether; butylbenzylphthalate; 1,2-, 1,3-, and 1,4-dichlorobenzene; alpha-endosulfan; beta-endosulfan; fluorene; hexachloroethane; methoxychlor; toxaphene; 1,2,4-trichlorobenzene; and m-xylene.
m-Xylene Tier II value applied to total xylene isomers.
Pentachlorophenol AWQC calculated to reflect site-specific pH (average in 1998 = 7.8).
DDT Tier II value applied to 4,4'-DDT.
PCBs Tier II value applied to sum of Aroclors®.
Trichloroethylene and tetrachloroethylene Tier II values applied to trichloroethene and tetrachloroethene.

USEPA Recommended Water Quality Criteria (USEPA, 1999c)
CCC - criteria continuous concentration
CMC - criteria maximum concentration
Pentachlorophenol criteria calculated to reflect site-specific pH (average in 1998 = 7.8).
Chlordane isomers not specified; criteria applied separately to each chlordane isomer.
Dieldrin CCC derivation did not consider dietary exposure.
Endosulfan criteria presented separately for alpha- and beta-endosulfan are more appropriately applied to the sum of these isomers.
Endrin CCC derivation did not consider dietary exposure.
Heptachlor epoxide CCC derived from data for heptachlor.
The CCC for PCBs applied to sum of Aroclors®.

USEPA Water Quality Criteria (1986a)
USEPA (1986a) values used only when USEPA (1999c) values unavailable.
Dichlorobenzenes criteria applied to sum of dichlorobenzene isomers.
Dichloropropane isomer not specified; criteria applied to 1,2-dichloropropane.
Trichlorinated ethanes criteria applied separately to trichloroethane isomers.
Tetrachlorinated ethanes acute criterion (USEPA 1986a) applied to 1,1,2,2-tetrachloroethane.
Dichloropropene isomers not specified; criteria applied separately to 1,3-dichloropropene isomers.
Trichloroethylene and tetrachloroethylene criteria applied to trichloroethene and tetrachloroethene.
Nitrophenols isomers not specified; criteria applied separately to nitrophenol isomers.
PCBs acute criterion applied to sum of Aroclors®.

Table 4-4. Ecological Screening Values Used for Metals in Onondaga Lake Surface Water[a]

Chemical	Measurement Basis	Conc. Units	USEPA Acute (Aquatic Life)	USEPA Chronic (Aquatic Life)	USEPA AWQC-FCV (Aquatic Life)	USEPA CCC (Aquatic Life)	USEPA CMC (Aquatic Life)	NYSDEC Acute (Aquatic)	NYSDEC Chronic (Aquatic)	USEPA Tier II (Aquatic Life)	NYSDEC (Wildlife)
Aluminum	dissolved	μg/L				87	750		100		
Aluminum	unfiltered	μg/L				87			100		
Antimony	dissolved	μg/L	9000	1600							
Arsenic	dissolved	μg/L			190	150	340	340	150		
Barium	dissolved	μg/L								3.9	
Beryllium	dissolved	μg/L	130	5.3					1,100	5.1	
Beryllium	unfiltered	μg/L							1,100		
Cadmium	dissolved	μg/L			1.3	2.8	5.9	5.4	2.6		
Chromium	dissolved	μg/L			228	94.8	729	729	94.8		
Cobalt	dissolved	μg/L							5	3	
Cobalt	unfiltered	μg/L							5		
Copper	dissolved	μg/L			14.7	11.6	17.8	17.8	11.6		
Iron	dissolved	μg/L			1,000	1,000		300	300		
Iron	unfiltered	μg/L				1,000		300	300		
Lead	dissolved	μg/L			3.7	3.5	89	134	5.2		
Manganese	dissolved	μg/L								80	
Methylmercury	dissolved	μg/L								0.003	
Mercury	dissolved	μg/L			1.3	0.77	1.4	1.4	0.77		0.0026
Nickel	dissolved	μg/L			203	67	604	604	67		
Selenium	dissolved	μg/L			5	4.6			4.6		
Selenium	unfiltered	μg/L									
Silver	dissolved	μg/L					5.8	6.8	0.1		
Silver	unfiltered	μg/L							0.1		
Thallium	dissolved	μg/L	1,400	40				20	8		
Thallium	unfiltered	μg/L						20	8		
Vanadium	dissolved	μg/L						190	14	19	
Vanadium	unfiltered	μg/L						190	14		
Zinc	dissolved	μg/L			135	152	151	151	107		
Cyanide	dissolved	μg/L			5.2	5.2	22	22	5.2		
Cyanide	unfiltered	μg/L				5.2	22	22	5.2		

[a] Cadmium, chromium, copper, lead, nickel, and zinc standards calculated using the lowest water hardness observed in 1992 (135 mg/L $CaCO_3$).

Table 4-4. (cont.)

Notes:

NYSDEC Water Quality Standards (NYSDEC, 1999a)

Arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc standards refer to dissolved fraction, and were compared only with dissolved concentrations. Standards for other metals were compared with both unfiltered and dissolved concentrations.

Ionic aluminum standard applied to aluminum.

Beryllium, cobalt, thallium, and vanadium standards refer to acid-soluble forms and were applied to both unfiltered and dissolved concentrations.

Mercury standard applied to total mercury.

Ionic silver standard applied to silver.

Free CN (sum of HCN and CN^-) standard applied to cyanide.

USEPAEcotox Surface Water Benchmarks (USEPA, 1996a)

AWQC/FCV - EPA chronic ambient water quality criteria or EPA-derived final chronic values (USEPA1986a, 1986b, 1987)

Tier II - Values calculated using Great Lakes Water Quality Initiative Tier II methodology (40 CFR 9, 122, 123, 131, and 132 [1995]).

AWQC/FCV and Tier II benchmarks for metals refer to total dissolved chemical (USEPA1996) and were compared only with dissolved concentrations.

Standards for metals were compared to both unfiltered and dissolved concentrations.

Inorganic mercury AWQC/FCV applied to total mercury.

USEPA Recommended Water Quality Criteria (USEPA, 1999c)

CCC - criteria continuous concentration

CMC - criteria maximum concentration

Criteria for metals (other than aluminum, iron, and cyanide) refer to dissolved fraction and were compared only with dissolved concentrations.

Standards for other metals were compared with both unfiltered and dissolved concentrations.

Total aluminum criteria applied to aluminum.

Total arsenic criteria applied to arsenic.

Total mercury criteria were derived for inorganic mercury, but applicable to total mercury. Mercury criteria will be underprotective "if a substantial portion of the mercury in the water column is methylmercury."

Total selenium CCC applied to selenium. CCC for dissolved fraction calculated using 0.922 conversion factor (multiplied by standard for total recoverable selenium).

Free cyanide criteria applied to cyanide.

USEPA Water Quality Criteria (USEPA,1986a)

USEPA (1986a) values used only when USEPA(1999c) values unavailable.

Criteria for metals assumed to refer to dissolved fraction.

Table 4-5. Dry-weight Basis Ecological Screening Values Used for Sediments in Onondaga Lake

Chemical	Units dry weight	NYSDEC LEL (Benthos)	NYSDEC SEL (Benthos)	OME NOEL (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	USEPA TEC (Benthos)	USEPA PEC (Benthos)	USEPA NEC (Benthos)	NOAA ERL (Benthos)
Total Metals and Cyanide										
Aluminum	mg/kg							58,030	73,160	
Antimony	mg/kg	2	25							
Arsenic	mg/kg	6	33		6	33	12.1	57	92.9	8.2
Barium	mg/kg									
Beryllium	mg/kg									
Cadmium	mg/kg	0.6	9		0.6	10	0.592	11.7	41.1	1.2
Calcium	mg/kg									
Chromium	mg/kg	26	110		26	110	56	159	312	81
Cobalt	mg/kg									
Copper	mg/kg	16	110		16	110	28	77.7	54.8	34
Iron	mg/kg	20,000	40,000		20,000	40,000				
Lead	mg/kg	31	110		31	250	34.2	396	68.7	47
Magnesium	mg/kg									
Manganese	mg/kg	460	1,100		460	1,100	1,673	1,080	819	
Mercury	ng/g	150	1,300		200	2,000				150
Nickel	mg/kg	16	50		16	75	39.6	38.5	37.9	21
Potassium	mg/kg									
Selenium	mg/kg									
Silver	mg/kg	1	2.2							
Sodium	mg/kg									
Thallium	mg/kg									
Vanadium	mg/kg									
Zinc	mg/kg	120	270		120	820	159	1,532	541	150
Cyanide	mg/kg									
Semivolatile Organic Compounds										
Chlorinated Aromatic Hydrocarbons										
Hexachlorobenzene	μg/kg			10	20					
Low Molecular Weight Polycyclic Aromatic Hydrocarbons										
Naphthalene	μg/kg						32.8	688	290	160
Acenaphthylene	μg/kg									
Acenaphthene	μg/kg									16

Table 4-5. (cont.)

Chemical	Units	NYSDEC LEL (Benthos)	NYSDEC SEL (Benthos)	OME NOEL (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	USEPA TEC (Benthos)	USEPA PEC (Benthos)	USEPA NEC (Benthos)	NOAA ERL (Benthos)
Fluorene	μg/kg						34.6	652	1,800	
Phenanthrene	μg/kg				560					240
Anthracene	μg/kg				220		31.6	548	1,700	
2-Methylnaphthalene	μg/kg									
High Molecular Weight Polycyclic Aromatic Hydrocarbons										
Fluoranthene	μg/kg				750		64.2	834	7,500	600
Pyrene	μg/kg				490		570	3,225	6,100	665
Benz[a]anthracene	μg/kg						260	4,200	3,500	
Chrysene	μg/kg				340		500	5,200	4,000	
Benzo[b]fluoranthene	μg/kg									
Benzo[k]fluoranthene	μg/kg				240					
Benzo[a]pyrene	μg/kg				370		350	394	440	430
Indeno[1,2,3-cd]pyrene	μg/kg				200		78	837	3,800	
Dibenz[a,h]anthracene	μg/kg							28.2	870	
Benzo[ghi]perylene	μg/kg				170		290	6,300	3,800	
PAH (total)	μg/kg									4,000
Pesticides/Polychlorinated Biphenyls										
α-Hexachlorocyclohexane	μg/kg				6					
β-Hexachlorocyclohexane	μg/kg				5					
δ-Hexachlorocyclohexane	μg/kg									
γ-Hexachlorocyclohexane	μg/kg			0.2	3					
Aldrin	μg/kg				2					
Chlordane	μg/kg			5	7					
Dieldrin	μg/kg			0.6	2					
Endrin	μg/kg			0.5	3					
Heptachlor	μg/kg			0.3						
Heptachlor epoxide	μg/kg				5					
4,4'-DDD	μg/kg				8					
4,4'-DDE	μg/kg				5					2.2
4,4'-DDT	μg/kg				8					1.6
DDT and metabolites (sum)	μg/kg				7					1.6
Toxaphene	μg/kg									1.6

Table 4-5. (cont.)

Chemical	Units	NYSDEC LEL (Benthos)	NYSDEC SEL (Benthos)	OME NOEL (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	USEPA TEC (Benthos)	USEPA PEC (Benthos)	USEPA NEC (Benthos)	NOAA ERL (Benthos)
Aroclor® 1016	μg/kg				7					
Aroclor® 1221	μg/kg									
Aroclor® 1232	μg/kg									
Aroclor® 1242	μg/kg									
Aroclor® 1248	μg/kg				30					
Aroclor® 1254	μg/kg				60					
Aroclor® 1260	μg/kg				5					
Polychlorinated biphenyls (sum)	μg/kg			10	70		31.6	245	194	23

Notes:

Only groups of compounds with screening values are listed here. Organic-carbon normalized screening values are listed in Table4-6.

NYSDEC Sediment Criteria (NYSDEC, 1999b)

LEL - lowest effect level

SEL - severe effect level

OME Sediment Benchmarks (Persaud et al., 1993)

NOEL - no effect level

LEL - lowest effect level

SEL - severe effect level

USEPA Sediment Benchmarks (USEPA, 1996b)

TEC - threshold effect concentration

PEC - probable effect concentration

NEC - high no-effect concentration

Total PCB TEC/PEC/NEC applied to sum of Aroclors®.

NOAA Sediment Benchmarks (Long et al., 1995)

ER-L - effects range-low

Arsenic-III ER-L applied to arsenic.

Chromium-III ER-L applied to chromium.

Inorganic mercury ER-L applied to total mercury.

DDT ER-L applied to 4,4'-DDTand total DDT and metabolites.

PCB ER-L applied to sum of Aroclors®.

Table 4-6. Organic-carbon Normalized Ecological Screening Values Used for Sediments in Onondaga Lake

	Units	NYSDEC Chronic (Benthos)	NYSDEC Acute (Benthos)	NYSDEC Bioaccumulation (Wildlife)	USEPA SQC* (Benthos)	USEPA SQB (Benthos)	OME LEL** (Benthos)	OME SEL (Benthos)	ORNL Secondary Chronic (Benthos)
Volatile Organic Compounds									
Aromatic Hydrocarbons									
Benzene	μg/gOC	28	103			5.7			16
Toluene	μg/gOC	49	235			67			5.0
Ethylbenzene	μg/gOC	24	212			360			8.9
Xylene isomers (total)	μg/gOC	92	833			2.5			16
Chlorinated Aromatic Hydrocarbons									
Chlorobenzene	μg/gOC	3.5	34.6			82			41
1,2-Dichlorobenzene	μg/gOC					34			33
1,3-Dichlorobenzene	μg/gOC					170			170
1,4-Dichlorobenzene	μg/gOC					35			34
Dichlorobenzenes (sum)	μg/gOC	12	120						
1,2,4-Trichlorobenzene	μg/gOC					920			960
Trichlorobenzenes (sum)	μg/gOC	91	910						
Halogenated Alkanes									
Methylene chloride	μg/gOC								37
1,1-Dichloroethane	μg/gOC								2.7
Chloroform	μg/gOC								2.2
1,2-Dichloroethane	μg/gOC								25
1,1,1-Trichloroethane	μg/gOC					17			3
Carbon tetrachloride	μg/gOC								4.7
1,1,2-Trichloroethane	μg/gOC								120
1,1,2,2-Tetrachloroethane	μg/gOC					94			140
Halogenated Alkenes									
1,1-Dichloroethene	μg/gOC								3.1
1,2-Dichloroethene isomers (total)	μg/gOC								40
cis-1,3-Dichloropropene	μg/gOC								0.0051
trans-1,3-Dichloropropene	μg/gOC								0.0051
Trichloroethene	μg/gOC					160			22
Tetrachloroethene	μg/gOC					53			41

Table 4-6. (cont.)

	Units	NYSDEC Chronic (Benthos)	NYSDEC Acute (Benthos)	NYSDEC Bioaccumulation (Wildlife)	USEPA SQC (Benthos)	USEPA SQB (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	ORNL Secondary Chronic (Benthos)
Ketones									
Acetone	μg/gOC								0.87
2-Butanone	μg/gOC								27
2-Hexanone	μg/gOC								2.2
4-Methyl-2-pentanone	μg/gOC								3.3
Miscellaneous Volatile Compounds									
Carbon disulfide	μg/gOC								0.085
Semivolatile Organic Compounds									
Chlorinated Aromatic Hydrocarbons									
Pentachlorobenzene	μg/gOC					69			70.1
Hexachlorobenzene	μg/gOC	5,570	9,081	12			2.0	24	
Low Molecular Weight Polycyclic Aromatic Hydrocarbons									
Naphthalene	μg/gOC	30	258			48			24
Acenaphthene	μg/gOC	140			62				130
Fluorene	μg/gOC	8	73			54	19	160	54
Phenanthrene	μg/gOC	120			85		56	950	180
Anthracene	μg/gOC	107	986				22	370	22
2-Methylnaphthalene	μg/gOC	34	304						
High Molecular Weight Polycyclic Aromatic Hydrocarbons									
Fluoranthene	μg/gOC	1,020			290		75	1,020	620
Pyrene	μg/gOC	961	8775				49	850	
Benz[a]anthracene	μg/gOC	12	94				32	1,480	11
Chrysene	μg/gOC						34	460	
Benzo[k]fluoranthene	μg/gOC						24	1,340	
Benzo[a]pyrene	μg/gOC						37	1,440	14
Indeno[1,2,3-cd]pyrene	μg/gOC						20	320	
Dibenz[a,h]anthracene	μg/gOC						6.0	130	
Benzo[ghi]perylene	μg/gOC						17	320	
Phenols									
Phenol	μg/gOC	0.5							3.1
2-Methylphenol	μg/gOC								1.2

Table 4-6. (cont.)

	Units	NYSDEC Chronic (Benthos)	NYSDEC Acute (Benthos)	NYSDEC Bioaccumulation (Wildlife)	USEPA SQC (Benthos)	USEPA SQB (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	ORNL Secondary Chronic (Benthos)
Substituted Phenols									
Pentachlorophenol	μg/gOC	40	100						
Chlorinated Aliphatic Hydrocarbons									
Hexachloroethane	μg/gOC					100			100
Hexachlorobutadiene	μg/gOC	5.5	55	4.0					
Hexachlorocyclopentadiene	μg/gOC	4.4	44						
Halogenated Ethers									
4-Bromophenyl-phenyl ether	μg/gOC					130			
Phthalates									
Diethyl phthalate	μg/gOC					63			60
Di-*n*-butyl phthalate	μg/gOC					1,100			1,100
Butylbenzyl phthalate	μg/gOC					1,100			1,100
Bis[2-ethylhexyl]phthalate	μg/gOC	199.5							89,000
Miscellaneous Oxygenated Compounds									
Dibenzofuran	μg/gOC					200			42
Pesticides/Polychlorinated Biphenyls									
α-Hexachlorocyclohexane	μg/gOC						0.6	10	12
β-Hexachlorocyclohexane	μg/gOC						0.5	21	12
δ-Hexachlorocyclohexane	μg/gOC								12
γ-Hexachlorocyclohexane	μg/gOC					0.37	0.3	1.0	0.37
Hexachlorocyclohexanes (sum)	μg/gOC	0.06	12.6	1.5					
Aldrin	μg/gOC						0.2	8.0	
α-Chlordane	μg/gOC	0.03	1.4	0.006			0.7	6.0	280
γ-Chlordane	μg/gOC	0.03	1.4	0.006			0.7	6.0	280
Dieldrin	μg/gOC	9.0			5.2		0.2	91	11
α-Endosulfan	μg/gOC	0.03	0.78			0.29			0.55
β-Endosulfan	μg/gOC	0.03	0.78			1.4			0.55
Endrin	μg/gOC	4.0		0.8	2.0		0.3	130	4.2
Heptachlor	μg/gOC								6.8
Heptachlor epoxide	μg/gOC						0.5	5.0	
Heptachlor and Heptachlor epoxide (sum)	μg/gOC	0.1	13.1	0.03					
Methoxychlor	μg/gOC	0.6				1.9			1.9

Table 4-6. (cont.)

	Units	NYSDEC Chronic (Benthos)	NYSDEC Acute (Benthos)	NYSDEC Bioaccumulation (Wildlife)	USEPA SQC (Benthos)	USEPA SQB (Benthos)	OME LEL (Benthos)	OME SEL (Benthos)	ORNL Secondary Chronic (Benthos)
4,4'-DDD	μg/gOC						0.8	6.0	11
4,4'-DDE	μg/gOC						0.5	19.0	
4,4'-DDT	μg/gOC	1.0	1,100				0.8	71	34
DDT and metabolites (sum)	μg/gOC						0.7	12	
Toxaphene	μg/gOC	0.01	3.2			2.8			
Aroclor® 1016	μg/gOC						0.7	53	
Aroclor® 1221	μg/gOC								12
Aroclor® 1232	μg/gOC								60
Aroclor® 1242	μg/gOC								17
Aroclor® 1248	μg/gOC						3.0	150	100
Aroclor® 1254	μg/gOC						6.0	34	81
Aroclor® 1260	μg/gOC						0.5	24	450,000
Polychlorinated biphenyls (sum)	μg/gOC	19.3	2,760	1.4			7.0	530	
Dioxins/Furans									
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin				0.0002					

Notes:

NYSDEC Sediment Criteria (NYSDEC, 1999b)

Xylene criteria applied to total xylene isomers.
Dichlorobenzenes criteria applied to sum of dichlorobenzene isomers.
Trichlorobenzenes criteria applied to sum of trichlorobenzene isomers.
Chlordane isomers not specified; criteria applied separately to each chlordane isomer.
Endosulfan isomers not specified; criteria applied separately to each endosulfan isomer.

Heptachlor and heptachlor epoxide criteria applied to sum of heptachlor and heptachlor epoxide.
Hexachlorocyclohexanes criteria applied to sum of hexachlorocyclohexane isomers.
PCB criteria applied to sum of Aroclors®.

Table 4-6. (cont.)

USEPA Sediment Benchmarks (USEPA, 1993a; 1995c)

SQC - sediment quality criteria (USEPA, 1993a)
SQB - sediment quality benchmarks (USEPA, 1995c)
SQB values derived by equilibrium partitioning.
m-Xylene SQB applied to total xylene isomers.
Trichloroethylene and tetrachloroethylene SQB values applied to trichloroethene and tetrachloroethene.
* Assumes 1% organic carbon.

OME Sediment Benchmarks (Persaud et al., 1993)

OME - Ontario Ministry of the Environment
LEL - lowest effect level (Persaud et al. 1993, except for PAHs [Persaud et al. 1991]) (assuming 1%TOC)
SEL - severe effect level (Persaud et al. 1993, except for PAHs [Persaud et al. 1991])
p,p'-DDD LEL/SEL and secondary chronic value applied to 4,4'-DDD.
p,p'-DDE LEL/SEL applied to 4,4'-DDE.
o,p'-DDT + p,p'-DDT LEL/SEL applied to 4,4'-DDT.
Total DDT LEL/SEL applied to sum of DDT and metabolites.
Chlordane isomers not specified for LEL/SEL or secondary chronic value; criteria applied separately to each chlordane isomer.
α-BHC LEL/SEL applied to α-hexachlorocyclohexane.
β-BHC LEL/SEL applied to β-hexachlorocyclohexane.
γ-BHC (lindane) LEL/SEL applied to γ-hexachlorocyclohexane.
Total PCB LEL/SEL applied to sum of Aroclors®.
** Assumes 1% organic carbon.

ORNL Secondary Chronic Benchmarks (Jones et al., 1997)

ORNL - Oak Ridge National Laboratory
NAWQC - national ambient water quality criterion
Secondary chronic benchmarks derived by equilibrium partitioning of aqueous benchmarks.
Xylene secondary chronic value applied to total xylene isomers.
1,2-Dichloroethene secondary chronic value applied to 1,2-dichloroethene isomers (total).
1,3-Dichloropropene isomers not specified for secondary chronic value; criterion applied separately to each 1,3-dichloropropene isomer.
Acenaphthene NAWQC chronic value used as secondary chronic benchmark.
Phenanthrene NAWQC chronic value used as secondary chronic benchmark.
Fluoranthene NAWQC chronic value used as secondary chronic benchmark.
Phenol NAWQC chronic value used as secondary chronic benchmark.
Chlordane NAWQC chronic value used as secondary chronic benchmark.
DDT secondary chronic value applied to 4,4'-DDT.
Dieldrin NAWQC chronic value used as secondary chronic benchmark.
Endosulfan, all isomers secondary chronic value applied separately to each endosulfan isomer.
Endrin NAWQC chronic value used as secondary chronic benchmark.
BHC (other) secondary chronic value applied separately to each hexachlorocyclohexane isomer (except lindane).
BHC (lindane) NAWQC chronic value used as secondary chronic benchmark for γ-hexachlorocyclohexane.

Table 4-7. Dry-weight Basis Ecological Screening Values Used for Soils Collected Near Onondaga Lake

Chemical	Units	Efroymson et al. 1997a Phytotoxicity	Efroymson et al. 1997b Microbial Toxicity	Efroymson et al. 1997b Earthworm Toxicity	USEPA Region 4 1999 Screening Value
Total Metals/Cyanide					
Aluminum	mg/kg	50	600		50 [a]
Antimony	mg/kg	5			3.5
Arsenic	mg/kg	10	100	60	10[a]
Barium	mg/kg	500	3,000		165
Beryllium	mg/kg	10			1.1
Boron	mg/kg	0.5	20		0.5 [a]
Cadmium	mg/kg	4	20	20	1.6
Calcium	mg/kg				
Chromium	mg/kg	1	10	0.4	0.4 [a]
Chromium VI	mg/kg				
Cobalt	mg/kg	20	1,000		20 [a]
Copper	mg/kg	100	100	50	40
Iron	mg/kg		200		200 [a]
Lead	mg/kg	50	900	500	50 [a]
Magnesium	mg/kg				
Manganese	mg/kg	500	100		100
Mercury	mg/kg	0.3 (inorganic)	30	0.1 (combined inorganic and organic)	0.1 (inorganic) [a]
Methylmercury	mg/kg				0.67
Molybdenum	mg/kg	2	200		2[a]
Nickel	mg/kg	30	90	200	30[a]
Potassium	mg/kg				
Selenium	mg/kg	1	100	70	0.81
Silver	mg/kg	2	50		2[a]
Sodium	mg/kg				
Thallium	mg/kg	1			1[a]
Vanadium	mg/kg	2	20		2[a]
Zinc	mg/kg	50	100	200	50[a]
Cyanide	mg/kg				0.9 (free total)
Total Cyanide	mg/kg				5.0 (total)

Table 4-7. (cont.)

Chemical	Units	Efroymson et al. 1997a Phytotoxicity	Efroymson et al. 1997b Microbial Toxicity	Efroymson et al. 1997b Earthworm Toxicity	USEPA Region 4 1999 Screening Value
Volatile Organic Compounds					
1,1,1-Trichloroethane	mg/kg				
1,1,2,2-Tetrachloroethane	mg/kg				
1,1-Dichloroethane	mg/kg				
1,2-Dichloroethane	mg/kg				0.4
1,2-Dichloroethene (Total)	mg/kg				
1,2-Dichloropropane	mg/kg			700	700 [a]
2-Butanone (MEK)	mg/kg				
4-Methyl-2-pentanone	mg/kg				
Acetone	mg/kg				
Benzene	mg/kg				0.05
Benzoic Acid	mg/kg				
BTX	mg/kg				
Carbon disulfide	mg/kg				
Chlorobenzene	mg/kg			40	0.05 (each); 0.05 (total)
Chloroethane	mg/kg				
Chloroform	mg/kg				
cis-1,2-Dichloroethene	mg/kg				
Ethylbenzene	mg/kg				0.05
Methylene chloride (Dichloromethane)	mg/kg				2.0
Pentachlorobenzene	mg/kg			20	0.0025
Tetrachloroethene	mg/kg				0.01
Toluene	mg/kg	200			0.05
Trichloroethene	mg/kg				
Vinyl chloride	mg/kg				0.01
Xylene (Total)	mg/kg				0.05
Xylene (m,p)	mg/kg				
Xylene (o)	mg/kg				
Semivolatile Organic Compounds					
Dichlorobenzenes (Total)	mg/kg				0.01
1,2-Dichlorobenzene	mg/kg				
1,3-Dichlorobenzene	mg/kg				
1,4-Dichlorobenzene	mg/kg			20	
1,2,3,4-Tetrachlorobenzene	mg/kg			10	0.01 (total)
Trichlorobenzenes (Total)	mg/kg				0.01
1,2,3-Trichlorobenzene	mg/kg			20	
1,2,4-Trichlorobenzene	mg/kg			20	
1,3,5-Trichlorobenzene	mg/kg				
1,2,4-Trimethylbenzene	mg/kg				

Table 4-7. (cont.)

Chemical	Units	Efroymson et al. 1997a Phytotoxicity	Efroymson et al. 1997b Microbial Toxicity	Efroymson et al. 1997b Earthworm Toxicity	USEPA Region 4 1999 Screening Value
Nitrobenzene	mg/kg			40	40 [a]
N-nitrosodiphenylamine	mg/kg			20	20 [a]
2-Methylnaphthalene	mg/kg				
2-Methylphenol	mg/kg				
4-Methylphenol	mg/kg				
2,4-Dichlorophenol	mg/kg				
2,4,5-Trichlorophenol	mg/kg	4			4
2,4,6-Trichlorophenol	mg/kg	10			10
4-Nitrophenol	mg/kg	7			7
2,4-Dinitrophenol	mg/kg	20			20
4-Chloroaniline	mg/kg				
Acenaphthene	mg/kg	20			20 [a]
Acenaphthylene	mg/kg	20			
Aniline	mg/kg				
Anthracene	mg/kg				0.1
Benz(a)anthracene	mg/kg				
Benzo(a)pyrene	mg/kg				0.1
Benzo(b)fluoranthene	mg/kg				
Benzo(g,h,i)perylene	mg/kg				
Benzo(k)fluoranthene	mg/kg				
Benzyl Alcohol	mg/kg				
Bis(2-ethylhexyl)phthalate	mg/kg				
sec-Butylbenzene	mg/kg				
Carbazole	mg/kg				
Chrysene	mg/kg				
Dibenz(a,h)anthracene	mg/kg				
Dibenzofuran	mg/kg				
Di-*n*-butyl phthalate	mg/kg	200			200 [a]
Diethylphthalate	mg/kg	100			100 [a]
Dimethylphthalate	mg/kg			200	200 [a]
Fluoranthene	mg/kg				0.1
Fluorene	mg/kg			30 (fluorine)	30 [a]
Hexachlorobenzene	mg/kg		1,000		0.0025
Hexachlorobutadiene	mg/kg				
Hexachloroethane	mg/kg				
Hexachlorocyclopentadiene	mg/kg	10			10
Indeno(1,2,3-cd)pyrene	mg/kg				
Isophorone	mg/kg				
Naphthalene	mg/kg				0.1
PAH Compounds	mg/kg				1.0 (total)
Pentachlorophenol	mg/kg	3	400	6	0.002
Phenanthrene	mg/kg				0.1
Phenol	mg/kg	70	100	30	0.05
Pyrene	mg/kg				0.1

Table 4-7. (cont.)

Chemical	Units	Efroymson et al. 1997a Phytotoxicity	Efroymson et al. 1997b Microbial Toxicity	Efroymson et al. 1997b Earthworm Toxicity	USEPA Region 4 1999 Screening Value
Pesticides/PCBs					
α-Hexachlorocyclohexane	mg/kg				0.0025
β-Hexachlorocyclohexane	mg/kg				0.001
Chlordane	mg/kg				
Aldrin	mg/kg				0.0025
Dieldrin	mg/kg				0.0005
Dieldrin & Aldrin	mg/kg				
Endosulfan	mg/kg				
Endrin	mg/kg				0.001
Heptachlor	mg/kg				
Heptachlor epoxide	mg/kg				
Methoxychlor	mg/kg				
4,4'-DDD	mg/kg				
4,4'-DDE	mg/kg				
4,4'-DDT	mg/kg				
Total DDT (Total, DDD,DDT,DDE)	mg/kg				0.0025
Aroclor-1016	mg/kg				
Aroclor-1221	mg/kg				
Aroclor-1232	mg/kg				
Aroclor-1242	mg/kg				
Aroclor-1248	mg/kg				
Aroclor-1254	mg/kg				
Aroclor-1254 & Aroclor 1260	mg/kg				
Aroclor-1260	mg/kg				
Total PCB	mg/kg	40			0.02
Dioxins/Furans					
Total PCDD/Fs	mg/kg				
Ecological Stressors					
Non-native species	mg/kg				
Wave disturbance	mg/kg				
Transparency	mg/kg				
Salinity	mg/kg				
Oncolites	mg/kg				
Conventional Analytes					
Ammonia	mg/kg				
Chloride	mg/kg				
Nitrite	mg/kg				
DO (mg/L)[3]	mg/kg				
pH (SU)	mg/kg				
Phosphorus	mg/kg				
Carbon tetrachloride	mg/kg		1,000		1,000
Sulfate	mg/kg				
Sulfides	mg/kg				

Note: [a]Based upon ORNL (Efroymson et al., 1997a; 1997b).

Table 4-8. ORNL Plant Screening Benchmarks

Analyte	Concentration Units	Measurement Basis	ORNL Soil Benchmark (Plants)
Total Metals and Cyanide			
Aluminum	mg/kg	dry	50
Antimony	mg/kg	dry	5
Arsenic	mg/kg	dry	10
Barium	mg/kg	dry	500
Beryllium	mg/kg	dry	10
Cadmium	mg/kg	dry	4
Chromium	mg/kg	dry	1
Cobalt	mg/kg	dry	20
Copper	mg/kg	dry	100
Lead	mg/kg	dry	50
Manganese	mg/kg	dry	500
Total mercury	mg/kg	dry	0.3
Nickel	mg/kg	dry	30
Selenium	mg/kg	dry	1
Silver	mg/kg	dry	2
Thallium	mg/kg	dry	1
Vanadium	mg/kg	dry	2
Zinc	mg/kg	dry	50
Volatile Organic Compounds			
Aromatic Hydrocarbons			
Styrene	μg/kg	dry	300,000
Toluene	μg/kg	dry	200,000
Semivolatile Organic Compounds			
Low Molecular Weight Polycyclic Aromatic Hydrocarbons			
Acenaphthene	μg/kg	dry	20,000
Phenols			
Phenol	μg/kg	dry	70,000
Substituted Phenols			
2,4-Dinitrophenol	μg/kg	dry	20,000
Pentachlorophenol	μg/kg	dry	3,000
2,4,5-Trichlorophenol	μg/kg	dry	4,000
Chlorinated Aliphatic Hydrocarbons			
Hexachlorocyclopentadiene	μg/kg	dry	10,000
Phthalates			
Di-*n*-butyl phthalate	μg/kg	dry	200,000
Di-ethyl phthalate	μg/kg	dry	100,000
Pesticides/Polychlorinated Biphenyls			
Polychlorinated biphenyls	μg/kg	dry	40,000

Note: ORNL - Oak Ridge National Laboratory
Source: Efroymson et al. (1997a).

Table 4-9. Fish Values for Screening-Level Exposure Estimates

Analyte	Concentration Units	Measurement Basis	Minimum Screening Value	Source of Minimum Screening Value
Total Metals and Cyanide				
Aluminum	mg/kg	wet	5.86	F
Antimony[1]	mg/kg	wet	0.38	F
Arsenic	mg/kg	wet	0.38	F
Barium	mg/kg	wet	30.2	F
Beryllium	mg/kg	wet	3.71	F
Cadmium	mg/kg	wet	2.86	E
Chromium	mg/kg	wet	1.97	E
Copper	mg/kg	wet	85.4	F
Lead	mg/kg	wet	2.23	E
Manganese	mg/kg	wet	494	F
Mercury (total)	mg/kg	wet	0.013	E
Nickel	mg/kg	wet	153	E
Selenium	mg/kg	wet	0.79	E
Thallium	mg/kg	wet	0.042	F
Vanadium	mg/kg	wet	1.1	F
Zinc	mg/kg	wet	28.6	E
Cyanide	mg/kg	wet	363	F
Volatile Organic Compounds				
Aromatic Hydrocarbons				
Benzene	μg/kg	wet	80,100	F
Toluene	μg/kg	wet	79,000	F
Xylene Isomers	μg/kg	wet	6,379	F
Halogenated Alkanes				
Methylene Chloride	μg/kg	wet	32,800	F
Chloroform	μg/kg	wet	84,000	F
1,2-Dichloroethane	μg/kg	wet	33,900	E
1,1,1-Trichloroethane	μg/kg	wet	3,157,000	F
Carbon tetrachloride	μg/kg	wet	89,800	F
Halogenated Alkenes				
Vinyl chloride	μg/kg	wet	954	F
Ketones				
Acetone	mg/kg	wet	56	F
2-Butanone	mg/kg	wet	9,943	F
4-Methyl-2-pentanone	mg/kg	wet	140	F
Semivolatile Organic Compounds				
Chlorinated Aromatic Hydrocarbons				
Hexachlorobenzene	μg/kg	wet	330	B
High Molecular Weight Polycyclic Aromatic Hydrocarbons				
Benzo[a]pyrene	μg/kg	wet	3,040	F
Phenols				
2-Methylphenol	μg/kg	wet	1,600,000	F
Substituted Phenols				
Tetrachlorophenol	μg/kg	wet	100	B
Pentachlorophenol	μg/kg	wet	1,347	F

Table 4-9. (cont.)

Analyte	Concen-tration Units	Measure-ment Basis	Minimum Screening Value	Source of Minimum Screening Value
Pesticides/Polychlorinated Biphenyls				
γ-Hexachlorocyclohexane	μg/kg	wet	3,950	E
Hexachlorocyclohexane isomers (Sum)	μg/kg	wet	100	B,F
Aldrin	μg/kg	wet	1,120	F
α-Chlordane	μg/kg	wet	4,200	E
γ-Chlordane	μg/kg	wet	4,200	E
Chlordane Isomers (Sum)	μg/kg	wet	500	B
Dieldrin	μg/kg	wet	110	F
Aldrin and Dieldrin (Sum)	μg/kg	wet	120	B
Endosulfan sulfate	μg/kg	wet	840	F
Endrin	μg/kg	wet	20	E
DDT and Metabolites (Sum)	μg/kg	wet	6	E
Aroclor® 1248	μg/kg	wet	109	F
Polychlorinated biphenyls	μg/kg	wet	100	A
Dioxins/Furans				
2,3,7,8-Tetrachlorodibenzofuran	ng/kg	wet	2	E
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	ng/kg	wet	5.6	F
1,2,3,7,8-Pentachlorodibenzofuran	ng/kg	wet	900	F
2,3,4,7,8-Pentachlorodibenzofuran	ng/kg	wet	90	F
1,2,3,6,7,8-Hexachlorodibenzofuran	ng/kg	wet	900	F

Notes:

A - IJC criteria (birds) (IJC, 1988)
B - NYSDEC criteria (piscivores) (Newell, 1987)
C - ORNL LOAEL (birds) (Sample et al., 1996)
[1] Arsenic value used to screen antimony

D - ORNL LOAEL (mammals) (Sample et al., 1996)
E - ORNL NOAEL (birds) (Sample et al., 1996)
F - ORNL NOAEL (mammals) (Sample et al., 1996)

Table 4-10. Avian Toxicity Reference Values for Screening-Level Exposure Estimates

Contaminant	Avian Toxicity Reference Value (mg/kg-day)	Reference
Total Metals and Cyanide		
Aluminum	110	Carriere et al. (1986)
Antimony	1,400	Damron and Wilson (1975)
Arsenic	2.46	USFWS (1969)
Barium	20.8	Johnson et al. (1960)
Beryllium	0.66	Schroeder and Mitchner (1975)*
Cadmium	1.45	White and Finley (1978)
Chromium	1	Haseltine et al. (unpublished data)
Cobalt	5	Nation et al. (1983)*
Copper	47	Mehring et al. (1960)
Lead	1.13	Edens et al. (1976)
Manganese	977	Laskey and Edens (1985)
Methylmercury	0.0064	Heinz (1979)
Mercury (total)	0.45	Hill and Schaffner (1976)
Nickel	77.4	Cain and Pafford (1981)
Selenium	0.4	Heinz et al. (1989)
Silver	18.1	Walker (1971)*
Thallium	0.237	Hudson et al. (1984)
Vanadium	11.4	White and Dieter (1978)
Zinc	14.5	Stahl et al. (1990)
Cyanide	68.7	Tewe and Maner (1981)*
Volatile Organic Compounds		
Aromatic Hydrocarbons		
Toluene	26	Nawrot and Staples (1979)*
Xylene (m,p)	2.1	Marks et al. (1982)*
Xylene (o)	2.1	Marks et al. (1982)*
Xylene isomers (total)	2.1	Marks et al. (1982)*
Chlorinated Aromatic Hydrocarbons		
1,2-Dichlorobenzene	6	Jori et al. (1982)
1,3-Dichlorobenzene	6	Jori et al. (1982)
1,4-Dichlorobenzene	6	Jori et al. (1982)
1,4-Dichlorobenzene	6	Jori et al. (1982)
Dichlorobenzenes (sum)	6	Jori et al. (1982)
1,2,4-Trichlorobenzene	8.0	Cote et al. (1988)*
1,2,3-Trichlorobenzene	8.0	Cote et al. (1988)*
1,3,5-Trichlorobenzene	8.0	Cote et al. (1988)*
Trichlorobenzenes (sum)	8.0	Cote et al. (1988)*
Halogenated Alkanes		
Methylene chloride	5.85	NCA (1982)*
Chloroform	15	Palmer et al. (1979)*
1,1-Dichloroethane	17.2	Alumut et al. (1976)[1]
1,2-Dichloroethane	17.2	Alumut et al. (1976)
1,1,1-Trichloroethane	1,000	Lane et al. (1982)*
1,1,2-Trichloroethane	1,000	Lane et al. (1982)*
Bromodichloromethane	N/A	
Carbon tetrachloride	16	Alumot et al. (1976)*

Table 4-10. (cont.)

Contaminant	Avian Toxicity Reference Value (mg/kg-day)	Reference
Halogenated Alkenes		
Vinyl chloride	0.17	Feron et al. (1981)*
1,1-Dichloroethene	2.5	Quast et al. (1983)*
cis -1,2-Dichloroethene	2.5	Quast et al. (1983)*
trans -1,2-Dichloroethene	2.5	Quast et al. (1983)*
1,2-Dichloroethene isomers (total)	2.5	Quast et al. (1983)*
Trichloroethene	0.7	Buben and O'Flaherty (1985)*
Tetrachloroethene	1.4	Buben and O'Flaherty (1985)*
Ketones		
Acetone	10	EPA (1986e)*
Semivolatile Organic Compounds		
Chlorinated Aromatic Hydrocarbons		
1,2,4,5-Tetrachlorobenzene	1.60	Grant et al. (1977)*
Tetrachlorobenzenes (mixed)	1.60	Grant et al. (1977)*
Pentachlorobenzene	6	Jori et al. (1982)
Hexachlorobenzene	0.2	Vos et al. (1971)
Low Molecular Weight Polycyclic Aromatic Hydrocarbons		
Naphthalene	1	Hough et al. (1993)[2]
Acenaphthylene	1	Hough et al. (1993)[2]
Acenaphthene	1	Hough et al. (1993)[2]
Fluorene	1	Hough et al. (1993)[2]
Phenanthrene	1	Hough et al. (1993)[2]
Anthracene	1	Hough et al. (1993)[2]
1-Methylnaphthalene	1	Hough et al. (1993)[2]
2-Methylnaphthalene	1	Hough et al. (1993)[2]
High Molecular Weight Polycyclic Aromatic Hydrocarbons		
Fluoranthene	1	Hough et al. (1993)[2]
Pyrene	1	Hough et al. (1993)[2]
Benz[a]anthracene	1	Hough et al. (1993)[2]
Chrysene	1	Hough et al. (1993)[2]
Benzo[b]fluoranthene	1	Hough et al. (1993)[2]
Benzo[k]fluoranthene	1	Hough et al. (1993)[2]
Benzo[a]pyrene	1	Hough et al. (1993)
Indeno[1,2,3-cd]pyrene	1	Hough et al. (1993)[2]
Dibenz[a,h]anthracene	1	Hough et al. (1993)[2]
Benzo[ghi]perylene	1	Hough et al. (1993)[2]
Benzo[e]pyrene	1	Hough et al. (1993)[2]
Phenols		
Phenol	6	Schafer et al. (1983)
2-Methylphenol	219	Hornshaw et al. (1986)*
4-Methylphenol	219	Hornshaw et al. (1986)*
Substituted Phenols		
Pentachlorophenol	0.24	Schwetz et al. (1978)*
2,3,4,6-Tetrachlorophenol	1	Hattula et al. (1981)*
Chlorinated Aliphatic Hydrocarbons		
Hexachloroethane	0.05	Tugarinova et al. (1960)*
Hexachlorobutadiene	0.2	Kociba et al. (1977)*

Table 4-10. (cont.)

Contaminant	Avian Toxicity Reference Value (mg/kg-day)	Reference
Phthalates		
Di-*n*-butyl phthalate	0.11	Peakall (1974)
bis[2-Ethylhexyl]phthalate	1.1	Peakall (1974)
Diethyl phthalate	4,580	Lamb et al. (1987)*
Pesticides/Polychlorinated Biphenyls		
α-Hexachlorocyclohexane	0.02	Sauter and Steele (1972)[3]
β-Hexachlorocyclohexane	0.02	Sauter and Steele (1972)[3]
δ-Hexachlorocyclohexane	0.02	Sauter and Steele (1972)[3]
γ-Hexachlorocyclohexane	0.02	Sauter and Steele (1972)[3]
Hexachlorocyclohexanes (sum)	0.02	Sauter and Steele (1972)
Aldrin	0.77	Mendenhall et al. (1983)
α-Chlordane	2.4	Stickel et al. (1983)
γ-Chlordane	2.4	Stickel et al. (1983)
Oxichlordane	2.4	Stickel et al. (1983)
Chlordane isomers (sum)	2.4	Stickel et al. (1983)
α-Endosulfan	50	Abiola (1992)
β-Endosulfan	50	Abiola (1992)
Endosulfan sulfate	11.1	Abiola (1992)
Dieldrin	0.077	Mendenhall et al. (1983)
Aldrin and dieldrin (sum)	0.77	Mendenhall et al. (1983)
Endrin	0.01	Fleming et al. (1982)
Endrin aldehyde	0.01	Fleming et al. (1982)[4]
Heptachlor	0.05	Wagstaff et al. (1980)
Heptachlor epoxide	0.005	WHO (1984)
Heptachlor and heptachlor epoxide (sum)	0.005	WHO (1984)
Methoxychlor	2	Hudson et al. (1984)
Mirex	20	Newell et al. (1987)
Photomirex	20	Newell et al. (1987)[5]
Mirex and photomirex	20	Newell et al. (1987)[5]
Toxaphene	8	Kennedy et al. (1973)*
4,4'-DDD	0.0028	Anderson et al. (1975)[5]
4,4'-DDE	0.038	Mendenhall et al. (1983)
4,4'-DDT	0.0028	Anderson et al. (1975)[6]
o,p'-DDD	0.0028	Anderson et al. (1975)[6]
o,p'-DDE	0.0028	Anderson et al. (1975)[6]
DDT and metabolites (sum)	0.0028	Anderson et al. (1975)
Aroclor® 1016	0.18	Dahlgren et al. (1982)
Aroclor® 1221	0.18	Dahlgren et al. (1982)
Aroclor® 1232	0.18	Dahlgren et al. (1982)
Aroclor® 1242	0.41	McLane and Hughes (1980)
Aroclor® 1248	0.18	Dahlgren et al. (1982)
Aroclor® 1254	0.18	Dahlgren et al. (1982)
Aroclor® 1260	33.3	Call and Harrell (1974)
Aroclor® 1268	0.18	Dahlgren et al. (1982)
Aroclor® 1254 and 1260	0.18	Dahlgren et al. (1982)
Aroclors (sum)	0.18	Dahlgren et al. (1982)

Table 4-10. (cont.)

Contaminant	Avian Toxicity Reference Value (mg/kg-day)	Reference
Dioxins/Furans		
2,3,7,8-Tetrachloro Dibenzodioxin (TCDD)	0.000014	Nosek et al. (1992)

Notes: Only compounds with TRVs available are listed here; all compounds can be found in screening tables (Appendix D).

* Indicates that a mammalian value was used due to the lack of an avian value.

[1] Value for 1,2-dichloroethane used.

[2] Value for benzo(a)pyrene used.

[3] Value for sum of hexachlorocyclohexane isomers used.

[4] Value for endrin used.

[5] Value for mirex used.

[6] Value for DDT and metabolites used.

Table 4-11. Mammalian Toxicity Reference Values for Screening-Level Exposure Estimates

Containinant	Mammalian Toxicity Reference Value (mg/kg-day)	Reference
Total Metals and Cyanide		
Aluminum	1.93	Ondreicka et al. (1966)
Antimony	0.125	Schroeder et al. (1968)
Arsenic	0.126	Schroeder and Mitchner (1971)
Barium	5.1	Perry et al. (1983)
Beryllium	0.66	Schroeder and Mitchner (1975)
Cadmium	1	Sutou et al. (1980)
Chromium	3.28	Mackenzie et al. (1958)
Cobalt	5	Nation et al. (1983)
Copper	11.7	Aulerich et al. (1982)
Lead	8	Azar et al. (1973)
Manganese	88	Laskey et al. (1982)
Methylmercury	0.015	Wobeser et al. (1976)
Mercury (total)	1	Aulerich et al. (1974)
Nickel	40	Ambrose et al. (1976)
Selenium	0.2	Rosenfeld and Beath (1954)
Silver	18.1	Walker (1971)
Thallium	0.0074	Formigli et al. (1986)
Vanadium	0.21	Domingo et al. (1986)
Zinc	160	Schlicker and Cox (1968)
Cyanide	68.7	Tewe and Maner (1981)
Volatile Organic Compounds		
Aromatic Hydrocarbons		
Benzene	26	Nawrot and Staples (1979)
Toluene	26	Nawrot and Staples (1979)
Ethylbenzene	26	Nawrot and Staples (1979)
Xylene (m,p)	2.1	Marks et al. (1982)
Xylene (o)	2.1	Marks et al. (1982)
Xylene isomers (total)	2.1	Marks et al. (1982)
Chlorinated Aromatic Hydrocarbons		
1,2-Dichlorobenzene	250	Lake et al. (1997)
1,3-Dichlorobenzene	250	Lake et al. (1997)
1,4-Dichlorobenzene	250	Lake et al. (1997)
Dichlorobenzenes (sum)	250	Lake et al. (1997)
1,2,4-Trichlorobenzene	8.0	Cote et al. (1988)
1,2,3-Trichlorobenzene	8.0	Cote et al. (1988)
1,3,5-Trichlorobenzene	8.0	Cote et al. (1988)
Trichlorobenzenes (sum)	8.0	Cote et al. (1988)
Halogenated Alkanes		
Methylene chloride	5.85	NCA (1982)
Chloroform	15	Palmer et al. (1979)
1,1-Dichloroethane	50	Lane et al. (1982)[1]
1,2-Dichloroethane	50	Lane et al. (1982)
1,1,1-Trichloroethane	1,000	Lane et al. (1982)
1,1,2-Trichloroethane	1,000	Lane et al. (1982)[1]
Carbon tetrachloride	16	Alumot et al. (1976)

Table 4-11. (cont.)

Contaminant	Mammalian Toxicity Reference Value (mg/kg-day)	Reference
Halogenated Alkenes		
Vinyl chloride	0.17	Feron et al. (1981)
1,1-Dichloroethene	2.5	Quast et al. (1983)
cis-1,2-Dichloroethene	2.5	Quast et al. (1983)[2]
trans-1,2-Dichloroethene	2.5	Quast et al. (1983)[2]
1,2-Dichloroethene isomers (total)	2.5	Quast et al. (1983)[2]
Trichloroethene	0.7	Buben and O'Flaherty (1985)
Tetrachloroethene	1.4	Buben and O'Flaherty (1985)
Ketones		
Acetone	10	USEPA (1986c)
Semivolatile Organic Compounds		
Chlorinated Aromatic Hydrocarbons		
1,2,4,5-Tetrachlorobenzene	1.6	Grant et al. (1977)
Tetrachlorobenzenes (mixed)	1.6	Grant et al. (1977)
Pentachlorobenzene	1.6	Grant et al. (1977)
Hexachlorobenzene	0.05	Fassbender et al. (1977)
Low Molecular Weight Polycyclic Aromatic Hydrocarbons		
Naphthalene	1	Mackenzie and Angevine (1981)[3]
Acenaphthylene	1	Mackenzie and Angevine (1981)[3]
Acenaphthene	1	Mackenzie and Angevine (1981)[3]
Fluorene	1	Mackenzie and Angevine (1981)[3]
Phenanthrene	1	Mackenzie and Angevine (1981)[3]
Anthracene	1	Mackenzie and Angevine (1981)[3]
1-Methylnaphthalene	1	Mackenzie and Angevine (1981)[3]
2-Methylnaphthalene	1	Mackenzie and Angevine (1981)[3]
High Molecular Weight Polycyclic Aromatic Hydrocarbons		
Fluoranthene	1	Mackenzie and Angevine (1981)[3]
Pyrene	1	Mackenzie and Angevine (1981)[3]
Benz[a]anthracene	1	Mackenzie and Angevine (1981)[3]
Chrysene	1	Mackenzie and Angevine (1981)[3]
Benzo[b]fluoranthene	1	Mackenzie and Angevine (1981)[3]
Benzo[k]fluoranthene	1	Mackenzie and Angevine (1981)[3]
Benzo[a]pyrene	1	Mackenzie and Angevine (1981)[3]
Indeno[1,2,3-cd]pyrene	1	Mackenzie and Angevine (1981)[3]
Dibenz[a,h]anthracene	1	Mackenzie and Angevine (1981)[3]
Benzo[ghi]perylene	1	Mackenzie and Angevine (1981)[3]
Benzo[e]pyrene	1	Mackenzie and Angevine (1981)[3]
Phenols		
Phenol	523	NCI (1980)
2-Methylphenol	219	Hornshaw et al. (1986)
4-Methylphenol	219	Hornshaw et al. (1986)
Substituted Phenols		
Pentachlorophenol	0.24	Schwetz et al. (1978)
2,3,4,6-Tetrachlorophenol	1	Hattula et al. (1981)
Chlorinated Aliphatic Hydrocarbons		
Hexachloroethane	0.05	Tugarinova et al. (1960)
Hexachlorobutadiene	0.2	Kociba et al. (1977)

Table 4-11. (cont.)

Contaminant	Mammalian Toxicity Reference Value (mg/kg-day)	Reference
Phthalates		
Di-n-butyl phthalate	550	Lamb et al. (1987)
Bis[2-ethylhexyl]phthalate	18.3	Lamb et al. (1987)
Diethyl phthalate	4,580	Lamb et al. (1987)
Pesticides/Polychlorinated Biphenyls		
α-Hexachlorocyclohexane	8	Palmer et al. (1978)
δ-Hexachlorocyclohexane	8	Palmer et al. (1978)
γ-Hexachlorocyclohexane	8	Palmer et al. (1978)
Hexachlorocyclohexanes (sum)	1.6	Grant et al. (1977)
Aldrin	0.2	Treon and Cleveland (1955)
α-Chlordane	0.075	FAO/WHO (1983)[4]
γ-Chlordane	0.075	FAO/WHO (1983)[4]
Chlordane isomers (sum)	0.075	FAO/WHO (1983)
α-Endosulfan	0.15	Dikshith et al. (1984)
β-Endosulfan	0.15	Dikshith et al. (1984)
Endosulfan sulfate	0.15	Dikshith et al. (1984)
Dieldrin	0.018	Harr et al. (1970)
Aldrin and dieldrin (sum)	0.018	Harr et al. (1970)
Endrin	0.065	Treon et al. (1955)
Endrin aldehyde	0.065	Treon et al. (1955)[5]
Endrin ketone	0.065	Treon et al. (1955)[5]
Heptachlor	0.075	Kinoshita and Kempf (1970)
Heptachlor epoxide	0.075	Kinoshita and Kempf (1970)
Heptachlor and Heptachlor epoxide	0.075	Kinoshita and Kempf (1970)[6]
Methoxychlor	4	Gray et al. (1988)
Toxaphene	8	Kennedy et al. (1973)
4,4'-DDD	85	NCI (1978)
4,4'-DDE	28	Gellert and Heinrich (1975)
4,4'-DDT	0.8	Fitzhugh (1948)
o,p'-DDD	0.8	Fitzhugh (1948)[7]
o,p'-DDE	0.8	Fitzhugh (1948)[7]
DDT and metabolites (sum)	0.8	Fitzhugh (1948)[7]
Mirex	0.05	Chu et al. (1981)
Photomirex	0.05	Chu et al. (1981)[8]
Mirex and photomirex	0.05	Chu et al. (1981)[8]
Aroclor® 1016	1.37	Bleavins et al. (1980)
Aroclor® 1221	0.01	Barsotti et al. (1976)
Aroclor® 1232	0.01	Barsotti et al. (1976)
Aroclor® 1242	0.069	Bleavins et al. (1980)
Aroclor® 1248	0.01	Barsotti et al. (1976)
Aroclor® 1254	0.068	McCoy et al. (1995)
Aroclors® 1254 and 1260	0.068	McCoy et al. (1995)
Aroclor® 1260	6.9	Linder et al. (1974)
Aroclor® 1268	0.01	Barsotti et al. (1976)
Aroclors (sum)	0.068	McCoy et al. (1995)

Table 4-11. (cont.)

Contaminant	Mammalian Toxicity Reference Value (mg/kg-day)	Reference
Dioxins/Furans		
2,3,7,8-TCDD	1.0E-06	Murray et al. (1979)

Notes: Only compounds with TRVs available are listed here; all compounds can be found in screening tables (Appe

[1] Value for 1,2-dichloroethane used.

[2] Value for 1,2-dichloroethane used.

[3] Value for benzo(a)pyrene used.

[4] Value for chlordane used.

[5] Value for endrin used.

[6] Value for heptaclor/heptaclor epoxide used.

[7] Value for DDT used.

[8] Value for mirex used.

5. Screening-Level Exposure Estimate and Risk Calculation (ERAGS Step 2)

The screening-level exposure estimate and risk calculation comprises the second step of ecological risk screening and was conducted consistent with USEPA guidance (USEPA, 1997a). Risk to receptors is estimated by comparing maximum documented exposure concentrations with the ecotoxicity screening values selected in Step 1 (Chapter 4, Section 4.2). The screening-level assessment serves to identify exposure pathways and contaminants of potential concern (COPCs) for the BERA by eliminating those contaminants and exposure pathways that pose negligible risks (USEPA, 1997a). These estimates ensure that the appropriate COCs are selected for further evaluation, and identifies data gaps for additional sampling or uncertainties to be addressed in the BERA.

5.1 Screening-Level Exposure Estimates

5.1.1 Ratios of Contaminants to Screening Criteria

The screening evaluation was conducted by comparing the maximum detected concentration, or half of the maximum detection limit, for each medium with the minimum (i.e., most conservative) screening criterion available (Chapter 3, Section 3.4). Screening values were selected based on availability and applicability to the site and to the freshwater environment and can be found in Chapter 4, Tables 4-3 to 4-9. The data used in the screening evaluations include the analytical results for surface water, sediment, soil, and fish tissue samples collected for the RI/FS by Honeywell in 1992, 1999, and 2000, and NYSDEC fish data collected between 1992 and 2000, and NYSDEC wetland data collected in 2002. A contaminant was selected for further examination when the ratio of the maximum detected concentration, or half of the detection level, to the minimum screening criterion equaled or exceeded 1.0 (unity). Food-web modeling was used to screen contaminants for the avian and mammalian receptors identified in Chapter 4, Section 4.1.2, as described below.

5.1.2 Food-Web Modeling

Screening-level exposure estimates were calculated for avian and mammalian receptors using the conservative exposure parameters listed in Tables 5-1 and 5-2 and the toxicity reference values (TRVs) selected in Chapter 4, Section 4.2 (Tables 4-10 and 4-11). The minimum adult weight found in the literature for each receptor species was used as the body weight.

Food ingestion rates (FIRs) were calculated in grams of dry matter per day using the following equations from Nagy (1987):

$$\text{FIR (g/day)} = 0.648\ \text{Wt.}^{0.651}\text{(g)} \qquad \text{all birds}$$

$$\text{FIR (g/day)} = 0.235\ \text{Wt.}^{0.822}\text{(g)} \qquad \text{all mammals}$$

Water ingestion rates (WIRs) were calculated using the following equations from Calder and Braun (1983):

$$\text{WIR (L/day)} = 0.059\ \text{Wt.}^{0.67}\text{(kg)} \qquad \text{all birds}$$

$$\text{WIR (L/day)} = 0.099\ \text{Wt.}^{0.90}\text{(kg)} \qquad \text{all mammals}$$

Sediment ingestion rates (SIRs) were based on Beyer et al. (1994), or professional judgment if a value was not available in Beyer for a species.

FIRs, WIRs, and SIRs are presented in Table 5-1 for birds and in Table 5-2 for mammals. All ingestion rates were divided by receptor body weights to provide intake rates per kg of body weight per day.

The general structure of the model used to estimate the exposure rate for a given chemical by a wildlife receptor is as follows:

$$\text{EED} = \sum\left(\text{IR}_p \times [\text{COC}]_p + \text{IR}_w \times [\text{COC}]_w + \text{IR}_s \times [\text{COC}]_s\right)$$

where:

EED	=	estimated environmental dose (mg/kg body weight-day)
IR_p (or FIR)	=	receptor-specific prey FIR (kg dry weight/kg body weight-day)
IR_w (or WIR)	=	receptor-specific WIR (L/kg body weight-day)
IR_s (or SIR)	=	receptor-specific incidental SIR (kg dry weight/kg body weight-day)
$[COC]_p$	=	COC concentration in the receptors' prey (mg/kg dry weight)
$[COC]_w$	=	COC concentration in the receptors' drinking water (mg/L)
$[COC]_s$	=	COC concentration in incidentally ingested sediments or soil (mg/kg dry weight)

The estimated environmental dose for each COC was divided by its TRV to calculate the hazard quotient (HQ).

5.2 Screening-Level Risk Calculations and Results

Appendix D presents the detailed results of the screening-level risk calculations conducted to identify potential COPCs for the BERA. The introduction to Appendix D contains a summary of the information contained within and the methods used in its calculations. Appendix E provides figures comparing detected levels of contaminants in Onondaga Lake surface sediments to NYSDEC sediment screening criteria (NYSDEC, 1999b).

Substances for which maximum detected site concentrations in surface water, sediment, soil, or fish tissue exceeded the lowest available screening values, or had HQs equal to or greater than 1.0, were considered COPCs and were retained for further evaluation in this BERA (Chapter 6, Section 6.1), as follows:

- Table 5-3 presents contaminants with screening ratios greater than 1.0 in Onondaga Lake surface water.
- Table 5-4 presents ratios greater than 1.0 in tributaries of Onondaga Lake for base, intermediate, and high flows. Since the tributaries of Onondaga Lake are not considered part of the site being evaluated by this BERA and therefore were not used to select COPCs, these data are provided for information only.
- Sediment screening ratios exceeding 1.0 are presented in Table 5-5.
- Soil screening ratios exceeding 1.0 are presented in Table 5-6.
- Plant screening ratios greater than 1.0 are presented in Table 5-7.
- Fish ratios greater than 1.0 are presented in Table 5-8.
- Food-web modeling results are summarized for avian and mammalian receptors in Table 5-9.
- Table 5-10 provides a summary of the contaminants exceeding HQs in the various media/receptors screened.

Based on the results of the screening-level ecological risk calculations summarized in Tables 5-3 to 5-10, it was determined that the contaminants from the site pose the risk of potential adverse effects. Therefore, the decision was made to continue with Steps 3 through 7 of the ecological risk assessment process (Chapters 6 through 11 of this report).

Table 5-1. Screening-Level Avian Receptor Life History Parameters

Factors	Units	Tree Swallow *Tachycineta bicolor*	Mallard Duck *Anas platyrhnchos*	Belted Kingfisher *Ceryle alcyon*	Great Blue Heron *Ardea herodias*	Osprey *Pandion haliaetus*	Red-Tailed Hawk *Buteo jamaicensis*
Body weight	kg	0.017 [a]	1.01 [b]	0.136 [c]	1.905 [d]	1.25 [e]	1.154 [f]
FIR (dw basis)	kg/kg-day	0.241	0.058	0.117	0.046	0.054	0.055
Percent dietary composition:							
Fish				100%	100%	100%	
Aquatic Invertebrates		100%	100%				
Plants							
Small mammals							100%
WIR	L/kg-day	0.23	0.059	0.114	0.048	0.055	0.056
SIR (dw basis)	% of FIR	0	3.3	1	2	0	2

Notes: dw – dry weight

FIR – food ingestion rate

SIR – sediment ingestion rate

WIR – water ingestion rate

Temporal and area use are assumed to be 100%.

FIR based on Nagy (1987) – FI (g/day) = 0.648 Wt.$^{0.651}$ (g).

WIR based on Calder and Braun (1983) – WI (L/day) = 0.059 Wt.$^{0.67}$ (g).

SIR based on Beyer et al. (1994) and/or professional judgment.

[a] Robertson et al. (1992).

[b] Dunning (1993).

[c] Brooks and Davis (1987).

[d] Poole (1938), as cited in USEPA (1993b).

[e] Brown and Amadon (1968).

[f] Steenhof (1983), as cited in USEPA (1993).

Table 5-2. Screening-Level Mammalian Receptor Life History Parameters

Factors	Units	Little Brown Bat *Myotis lucifugus*	Short-Tailed Shrew *Blarina brevida*	Mink *Mustela vison*	River Otter *Lutra canadensis*
Body weight	kg	0.0045 [a]	0.015 [b]	0.55 [c]	4.74 [d]
FIR (dw basis)	kg/kg-day	0.180	0.145	0.076	0.052
Percent dietary composition –					
Fish				100%	100%
Aquatic invertebrates (emergent)		100%			
Terrestrial invertebrates			100%		
WIR	L/kg-day	0.170	0.151	0.105	0.085
SIR (dw basis)	% of FIR	0	13[e]	1	1

Notes: dw – dry weight
FIR – food ingestion rate
SIR – sediment ingestion rate
WIR – water ingestion rate
Temporal and area use are assumed to be 100%.
FIR based on Nagy (1987) – FI (g/day) = 0.235 Wt. $^{0.822}$ (g).
WIR based on Calder and Braun (1983) – WI (L/day) = 0.099 Wt. $^{0.90}$ (g).
Professional judgment used to estimate SIR when no reference was available.
[a] Doutt et al. (1977).
[b] Schlesinger and Potter (1974).
[c] Mitchell (1961).
[d] Lauhachinda (1974), as cited in USEPA (1993b).
[e] Sample and Suter (1994).

Table 5-3. Summary of Screening Ratios that Exceeded 1.0 for Surface Water in Onondaga Lake in 1992 and 1999

Chemical	Screening Ratio[a] 1992	1999	1997-2001*
Conventional Analytes			
Chloride	2.4		
Dissolved Oxygen			O_2 below 4.0 mg/L
Sulfide			11,000
Metals and Cyanide			
Aluminum	1.4		
Barium	20		
Cadmium	2.2		
Copper	4.4		
Iron	1.2	2.0	
Lead	5.2	<1.0	
Manganese	11	7.6	
Methylmercury	4.1	5.1	
Total Mercury	23	40	
Zinc	1.3		
Cyanide	33		
Volatile Organic Compounds			
Xylenes	2.8[b]	<1.0	
Chlorobenzene	1.0[b]	2.4	
Dichlorobenzenes (Sum)		1.3	
Trichlorobenzenes (Sum)	1.3		
Semivolatile Organic Compounds			
bis[2-Ethylhexyl]phthalate	17		
Hexachlorobenzene	1.4[b]		
Fluorene	1.3[b]		
Benzo[a]pyrene	357[b]		
Pentachlorophenol	1.0[b]		
Hexachlorobutadiene	5.0[b]		
Hexachlorocyclopentadiene	11[b]		
4-Bromophenyl-phenyl ether	3.3[b]		
Pesticides/Polychlorinated Biphenyls			
α-Chlordane	6.0[b]		
γ-Chlordane	6.0[b]		
α-Endosulfan	2.9[b]		
β-Endosulfan	5.6[b]		
Endrin	1.4[b]		
Heptachlor	6.8[b]		
Heptachlor epoxide	6.8[b]		
Methoxychlor	14[b]		
4,4'-DDT	50[b]		
Toxaphene	13,000[b]		

Notes: [a] Ratios are maximum detected values divided by minimum screening values. When two values were available (e.g., dissolved and unfiltered), the higher value was used.

[b] Ratio is halved maximum detection limit divided by the minimum screening value. Full TCL/TAL analyses were not performed in 1999.

* OCDWEP 1997 to 2001 Data

Table 5-4. Summary of Screening Ratios that Exceeded 1.0 for Onondaga Lake Tributary Water and Metro Discharge in 1992

Chemical	Screening Ratio[a] Base Flow	Intermediate Flow	High Flow
Conventional Analytes			
Chloride	4.8	4.5	4.3
Metals			
Cadmium	<1.0[b]	1.8	13
Chromium	<1.0	1.3	5.9
Copper	1.3	2.6	11
Iron	1.9	N/A	N/A
Lead	3.1	8.1	27
Manganese	1.5	1.2	N/A
Methylmercury	1.0	1	3.1
Total Mercury	84	159	145
Nickel	1.7	1.4	4.9
Zinc	1.8	1.7	2.4
Volatile Organic Compounds			
Benzene	<1.0	<1.0[b]	1.3
Xylenes	2.0	<1.0[b]	1.3
Chlorobenzene	1.5	<1.0[b]	<1.0[b]
1,4-Dichlorobenzene	<1.0	<1.0	1.4
Dichlorobenzenes (Sum)	<1.0	<1.0	4.5

Notes: Tributaries sampled include: Bloody Brook, East Flume, Geddes Brook, Harbor Brook, Lake Outlet, Ley Creek, Ninemile Creek, Onondaga Creek, and Sawmill Creek.

Highest concentration detected in all tributaries is presented.

N/A – not available.

[a] Ratios are maximum detected values (unfiltered) divided by minimum screening values.

[b] Ratio is halved maximum detection limit divided by the minimum screening value.

Table 5-5. Summary of Screening Ratios that Exceeded 1.0 for Sediments in Onondaga Lake in 1992 and 2000[a]

Chemical	Screening Ratio[b] 1992	2000
Metals		
Antimony	3.2	2.7
Arsenic	1.9	7.9
Cadmium	24	25
Chromium	77	161
Copper	11	23
Iron	1.7	2.5
Lead	8.1	24
Manganese	1.1	2.6
Mercury	459	518
Nickel	41	104
Silver	5.1	6.1
Zinc	2.3	3.5
Volatile Organic Compounds		
Benzene[d]	154	246
Toluene[d]	52	28
Ethylbenzene[d]	11	26
Xylenes[d]	400	825
Chlorobenzene[d]	657	4,571
Dichlorobenzenes (Sum)[d]	106	230
1,2-Dichlorobenzene[d]	15	24
1,3-Dichlorobenzene[d]	2.1	1.9
1,4-Dichlorobenzene[d]	29	59
Trichlorobenzenes (Sum)[d]	2.9	3.3
Methylene chloride[d]	<1.0	3.0
1,1-Dichloroethane[d]	44[c]	43[c]
Chloroform[d]	55[c]	52[c]
1,2-Dichloroethane[d]	4.8[c]	<1.0
1,1,1-Trichloroethane[d]	40[c]	<1.0
Carbon tetrachloride[d]	26[c]	24[c]
1,1,2-Trichloroethane[d]	1.0[c]	<1.0[c]
1,1,2,2-Tetrachloroethane[d]	1.3[c]	1.2[c]
1,1-Dichloroethene[d]	39[c]	37[c]
1,2-Dichloroethene isomers (total)[d]	3.0[c]	N/A
cis-1,3-Dichloropropene[d]	23,529[c]	22,549[c]
trans-1,3-Dichloropropene[d]	23,529[c]	22,549[c]
Trichloroethene[d]	5.5[c]	<1.0[c]
Tetrachloroethene[d]	2.9[c]	<1.0
Acetone[d]	44	7.0
2-Butanone[d]	8.1	<1.0
2-Hexanone[d]	55[c]	108[c]
4-Methyl-2-pentanone[d]	36[c]	N/A
Carbon disulfide[u]	1,412[c]	18

Table 5-5. (cont.)

Chemical	Screening Ratio[b]	
	1992	2000
Semivolatile Organic Compounds		
Hexachlorobenzene[d]	55	22
Naphthalene	915	792,683
Acenaphthene	938	5,313
Fluorene	208	4,046
Phenanthrene	92	2,625
Anthracene	282	3,006
2-Methylnaphthalene[d]	8.4	262
Fluoranthene	3,583	3,894
Pyrene[d]	41	306
Benz[a]anthracene[d]	83	385
Chrysene[d]	29	294
Benzo[k]fluoranthene[d]	22	18
Benzo[a]pyrene[d]	49	186
Indeno[1,2,3-cd]pyrene	25	487
Dibenz[a,h]anthracene	32	603
Benzo[g,h,i]perylene[d]	45	121
Phenol[d]	18	106
2-Methylphenol[d]	1,083[c]	12
Pentachlorophenol[d]	78[c]	239[c]
4-Chloro-3-methylphenol[d]	N/A	579[c]
Hexachloroethane[d]	13[c]	3.7[c]
Hexachlorobutadiene[d]	325[c]	92[c]
Hexachlorocyclopentadiene[d]	295[c]	434[c]
4-Bromophenyl-phenyl ether[d]	10[c]	2.8[c]
Diethyl phthalate[d]	22[c]	6.1[c]
Di-*n*-butyl phthalate[d]	1.2[c]	<1.0
Butylbenzyl phthalate[d]	1.2[c]	<1.0[c]
Dibenzofuran[d]	2.2	14
Pesticides/Polychlorinated Biphenyls		
β-Hexachlorocyclohexane[d]	1.0[c]	<1.0
γ-Hexachlorocyclohexane[d]	1.7[c]	<1.0
Hexachlorocyclohexanes[d]	N/A	2.2
Aldrin[d]	2.5[c]	<1.0
α-Chlordane[d]	67	28
γ-Chlordane[d]	83[c]	73
Dieldrin[d]	4.8[c]	1.8
α-Endosulfan[d]	17[c]	3.3
β-Endosulfan[d]	32[c]	1.6[c]
Endrin[d]	3.2[c]	<1.0
Toxaphene[d]	5,000[c]	474[c]
Heptachlor epoxide[d]	1.0[c]	<1.0
Heptachlor and heptachlor epoxide (Sum)[d]	N/A	11
Methoxychlor[d]	8.3[c]	<1.0

Table 5-5. (cont.)

Chemical	Screening Ratio[b] 1992	2000
DDT and metabolites[d]	5.1	1.4
4,4'-DDD[d]	4.5	<1.0
4,4'-DDE[d]	1.9[c]	1.0
4,4'-DDT	9.1[c]	55
PCBs (Sum)	210	2,096
Aroclor 1016[d]	14	7.7[c]
Aroclor 1221[d]	1.6[c]	1.7
Aroclor 1242[d]	<1.0[c]	6.8
Aroclor 1248[d]	20	<1.0
Aroclor 1254[d]	1.5	2.4
Aroclor 1260[d]	98	15
Dioxins/Furans		
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin[d]	N/A	11

Notes: [a] Maps of exceedances of NYSDEC sediment screening criteria are presented in Appendix E.

[b] Ratios are maximum detected values divided by minimum screening values.

[c] Ratio is halved maximum detection limit divided by the minimum screening value.

[d] Ratio is based on organic-carbon normalized values.

N/A = Not available.

Table 5-6. Summary of Screening Ratios that Exceeded 1.0 for Soil near Onondaga Lake in 2000 and 2002

Chemical	Screening Ratio[a]		
	Wetlands	Wetlands	Dredge Spoils Area
	(Soil Benchmarks)	*(Sediment Benchmarks)**	*(Soil Benchmarks)*
Metals and Cyanide			
Aluminum	274	<1.0	466
Arsenic	1.8	3.1	1.3
Barium	2.4	*	1.4
Beryllium	1.1	*	<1.0
Cadmium	8.9	24	2.7
Chromium	385	5.9	154
Copper	4.2	10	1.5
Iron	162	1.6	148
Lead	5.2	8.4	3.1
Manganese	4.9	1.1	4.4
Mercury	251	167	988
Nickel	2.5	4.7	1.7
Selenium	3.1	*	2.6
Silver	1.4	2.7	<1.0
Thallium	2.5	*	<1.0
Vanadium	18	*	20
Zinc	10	4.3	3.7
Cyanide	6.0	*	1.4
Volatile Organic Compounds			
Benzene	1.2	*	NA
Chlorobenzene	12	*	NA
1,2-Dichlorobenzene	490	*	9.7
1,3-Dichlorobenzene	138	*	13
1,4-Dichlorobenzene	490	*	13[b]
1,2,4-Trichlorobenzene	328	*	13[b]
Semivolatile Organic Compounds			
Hexachlorobenzene	2,142	536	196
Naphthalene	76	232	90
Phenanthrene	630	263	95
Anthracene	180	570	19
Fluoranthene	860	1,340	320
Pyrene	830	169	250
Benzo[a]pyrene	480	137	230
Phenol	57	*	5[b]
2-Chlorophenol	315[b]	*	25[b]
Pentachlorophenol	5,250[b]	*	3,250[b]
4-Nitrophenol	1.5[b]	*	<1.0
Pesticides/Polychlorinated Biphenyls			
β-Hexachlorocyclohexane	5.6	1.1	3.2
DDT and metabolites (Sum)	22	35	<1.0
Chlordane (Sum)	NA	5.9	NA
Aldrin	18	23	<1.0
Dieldrin	48	40	7.6
Total PCBs	54	107	41

Notes: *Only sediment benchmarks measured on a dry-weight basis were applied (see Table 4-5).
NA - denotes not analyzed
[a] Ratios are maximum detected values divided by minimum screening values.
[b] Ratio is halved maximum detection limit divided by the minimum screening value.

Table 5-7. Summary of Screening Ratios that Exceeded 1.0 for Plants near Onondaga Lake in 2000

Chemical	Screening Ratio[a]	
	Wetlands	Dredge Spoils Area
Aluminum	274	466
Arsenic	1.8	1.3
Cadmium	3.6	1.1
Chromium	154	62
Copper	1.7	<1.0
Lead	5.2	3.1
Manganese	1.0	<1.0
Mercury (Inorganic)	84	329
Nickel	2.1	1.7
Selenium	2.5	2.1
Silver	1.4	<1.0
Thallium	2.5	<1.0
Vanadium	18	20
Zinc	10	3.7

Note: [a] Ratios are maximum detected values divided by minimum screening values.

Table 5-8. Summary of Screening Ratios that Exceeded 1.0 for Fish in Onondaga Lake from 1992 to 2000

	Fish	
Chemical	**1992–1999**	**2000**
Metals		
Aluminum	0.8	20
Antimony	5.5	0.07
Arsenic	0.3	5.3
Chromium	<1.0	7.2
Mercury/methylmercury	390/244	145/NA
Selenium	1.1	3
Thallium	2.5[a]	<1.0
Vanadium	0.2	1.5
Zinc	<1.0	15
Organic Compounds		
Bis[2-ethylhexyl]phthalate	1.1	NA
Aroclor 1248	22	19
Aroclor 1254	31	0.1[a]
Polychlorinated biphenyl (sum)	110	34
DDT and metabolites	100	83
Oxichlordane	10[a]	2.4E-03
Mirex	3[a]	0.03
Photomirex	7.6[a]	7.6E-03
Endrin	1.6	2.3
γ-Hexachlorocyclohexane	2.3	2.5E-03
2,3,4,7,8-Pentachlorodibenzofuran	<1.0	1.3
2,3,7,8-Tetrachlorodibenzofuran	20	32

Notes: Higher ratio of adult and juvenile fish is presented.

[a] Ratio is halved maximum detection limit divided by the minimum screening value.

NA= not analyzed

Table 5-9. Results of Screening Risk Assessment for Detected Chemicals of Potential Concern through Food Web Exposure for Wildlife Receptors Using Maximum Concentrations (Hazard Quotients Greater than 1.0)

	Tree Swallow		Mallard		Belted Kingfisher		Great Blue Heron		Osprey	
Chemical	**1992–1998**	**1999–2000**	**1992–1998**	**1999–2000**	**1992–1998**	**1999–2000**	**1992–1998**	**1999–2000**	**1992–1998**	**1999–2000**
Metals										
Aluminum	7.2	19	2.6	4.7						
Antimony										
Arsenic		3.1								
Barium	95	222	23	54						
Cadmium	22	23	5.3	5.5						
Chromium	282	592	72	151	2.7	12	2.0	6.6		3.3
Cobalt	13	26	3.2	6.3						
Copper	5.0	7.3		1.8						
Lead	51	151	13	38		1.4				
Manganese										
Methylmercury	73	95	38	26	222	161	87	64	102	73
Mercury (total)	2.7	28		7.2	5.2	2.5	2.1	1.0	2.3	1.0
Nickel	6.4	16	1.6	4.0						
Selenium		2.1			1.0	3.4		1.4		1.6
Thallium		2.8								
Vanadium	1.4	4.6		1.1						
Zinc	39	59	9.4	14		10		4.1		4.7
Volatile Organic Compounds										
1,2-Dichlorobenzene	2.1	12		3.0						
1,3-Dichlorobenzene	1.8	10		2.4						
1,4-Dichlorobenzene	4.2	45	1.0	11						
Dichlorobenzene (sum)	6.0		1.5							
1,2,4-Trichlorobenzene		7.5		1.8						
Xylene (m,p)		17		4.1						
Xylene (o)		3.8								
Xylene Isomers		55		14						
Vinyl Chloride	3.5[a]	15[a]		3.8[a]						

Table 5-9. (cont.)

Chemical	Tree Swallow 1992–1998	Tree Swallow 1999–2000	Mallard 1992–1998	Mallard 1999–2000	Belted Kingfisher 1992–1998	Belted Kingfisher 1999–2000	Great Blue Heron 1992–1998	Great Blue Heron 1999–2000	Osprey 1992–1998	Osprey 1999–2000
Semivolatile Organic Compounds										
Pentachlorophenol	73[a]	4,368[a]	18[a]	1,075[a]		15[a]		12[a]		
Di-*n*-butyl phthalate	67[a]		16[a]							
Bis[2-ethylhexyl]phthalate	1.0	1.6			1.0					
Acenaphthylene	1.7	5.1		1.3						
Acenaphthene		3.2								
Anthracene		3.6		1.0						
Benz[a]anthracene		12		3.0						
Benzo[a]pyrene		5.1		1.3						
Benzo[g,h,i]perylene										
Chrysene	1.1	15		3.9						
Dibenz[a,h]anthracene		2.2								
Fluoranthene	43	47	11	12						
Benzo[b]fluoranthene	4.2	8.3	1.1	2.1						
Benzo[k]fluoranthene		6.1		1.6						
Fluorene	2.4	47		12						
Hexachlorobenzene	8.7	49	2.1	12						
Indeno[1,2,3-cd]pyrene		3.7		1.0						
1-Methylnaphthalene	15		3.7							
2-Methylnaphthalene	1.3	978		241		3.3		2.6		
Naphthalene	28	23,959	6.7	5,816		30		24		
Phenanthrene	2.3	67		17						
Pyrene	3.1	27		6.8						
Hexachloroethane	147[a]	804[a]	36[a]	198[a]	1.3[a]	2.7[a]		2.1[b]		
Hexachlorobutadiene	37[a]	201[a]	9[a]	49[a]						

Table 5-9. (cont.)

Chemical	Tree Swallow 1992–1998	Tree Swallow 1999–2000	Mallard 1992–1998	Mallard 1999–2000	Belted Kingfisher 1992–1998	Belted Kingfisher 1999–2000	Great Blue Heron 1992–1998	Great Blue Heron 1999–2000	Osprey 1992–1998	Osprey 1999–2000
Pesticides/PCBs										
γ-Hexachlorocyclohexane					201		79		93	
4,4'-DDD	2.4				19	15	7.6	6.0	8.9	7.1
4,4'-DDE					3.3	4.3	1.3	1.7	1.5	2.0
4,4'-DDT		4.5		1.1	12	7.4	4.6	2.9	5.3	3.4
DDT and metabolites	2.4				113	81	44	32	52	38
Chlordane isomers (sum)										
Heptachlor epoxide					1.1					
Heptachlor and heptachlor epoxide (sum)					1.1					
Dieldrin										
Endrin					1.7	2.1				1.0
Aroclor 1016		2.2[a]			4.0		1.6		1.8	
Aroclor 1221		4.5		1.1						
Aroclor 1232		2.2[a]								
Aroclor 1242		24		5.8	1.9					
Aroclor 1248	3.2				6.6	5.5	2.6	2.2	3.0	2.5
Aroclor 1254		3.6			30		12		14	
Aroclor 1254 and 1260					4.2	1.3	1.7		1.9	
Aroclor 1268										
PCBs (sum)	6.0	60	1.5	15	30	1.5	12	3.5	14	4.0
Dioxins/Furans										
Dioxins/furans (TEQ)		588		1.7	2.9	1.7			1.4	

Table 5-9. (cont.)

	Red-Tailed Hawk	Little Brown Bat		Short-Tailed Shrew	Mink		River Otter	
Chemical	**1999–2000**	**1992–1998**	**1999–2000**	**1999–2000**	**1992–1998**	**1999–2000**	**1992–1998**	**1999–2000**
Metals								
Aluminum	1.3	308	795	434	3.0	20	2.0	14
Antimony		6.3	5.3	1.7	5.3		3.6	
Arsenic		11	46	14		4.4		3.0
Barium		290	675	3.2				
Cadmium		24	25	57				
Chromium	2.3	64	135	22		2.4		1.6
Cobalt		10	19					
Copper		10	22	1.1				
Lead	2.0	5.3	16	7.8				
Manganese		1.0	2.3					
Methylmercury	180	23	30	138	61	45	42	30
Mercury (total)	2.6		9.6	2.1	1.5		1.0	
Nickel		9.2	24	1.1				
Selenium			3.1	3.4	1.3	4.5		3.1
Thallium		15	67	81	3.1	1.1	2.1	
Vanadium		58	185	6.1		3.6		2.5
Zinc		2.6	4.0	2.0				
Volatile Organic Compounds								
1,2-Dichlorobenzene								
1,3-Dichlorobenzene								
1,4-Dichlorobenzene								
Dichlorobenzene (sum)								
1,2,4-Trichlorobenzene			5.6	2.7				
Xylene (m,p)			12					
Xylene (o)			2.9					
Xylene Isomers		1.6	41					
Vinyl Chloride		2.6[a]	12[a]					

Table 5-9. (cont.)

Chemical	Red-Tailed Hawk	Little Brown Bat		Short-Tailed Shrew	Mink		River Otter	
	1999–2000	1992–1998	1999–2000	1999–2000	1992–1998	1999–2000	1992–1998	1999–2000
Semivolatile Organic Compounds								
Pentachlorophenol	5.3[a]	54[a]	3,263[a]	142[a]		9.5[a]		6.5[a]
Di-*n*-butyl phthalate								
Bis[2-ethylhexyl]phthalate								
Acenaphthylene		1.6	3.8	24				
Acenaphthene			2.4	20				
Anthracene	2.2		2.7	58				
Benz[a]anthracene	6.0		8.8	159				
Benzo[a]pyrene	5.9		3.8	155				
Benzo[ghi]perylene	3.4			91				
Chrysene	5.9		11	155				
Dibenz[a,h]anthracene	1.0		1.7	26				
Fluoranthene	11	32	35	278		0.2		
Benzo[b]fluoranthene	7.2	3.1	6.2	191				
Benzo[k]fluoranthene	2.7		4.6	71				
Fluorene	1.5	1.8	35	39				
Hexachlorobenzene	3.3	26	147	347		1.2		
Indeno[1,2,3-cd]pyrene	3.4		2.8	91				
1-Methylnaphthalene		11						
2-Methylnaphthalene		1.0	731	18		2.1		1.5
Naphthalene	1.1	21	17,895	29		20		14
Phenanthrene	7.7	1.7	50	204				
Pyrene	10	2.3	20	269				
Hexachloroethane	1.0[a]		600[a]	27[a]		1.7[a]		1.2[a]
Hexachlorobutadiene			150[a]	6.7[a]				

Table 5-9. (cont.)

Chemical	Red-Tailed Hawk	Little Brown Bat		Short-Tailed Shrew	Mink		River Otter	
	1999–2000	1992–1998	1999–2000	1999–2000	1992–1998	1999–2000	1992–1998	1999–2000
Pesticides/PCBs								
g-Hexachlorocyclohexane								
4,4'-DDD								
4,4'-DDE								
4,4'-DDT	2.2							
DDT and metabolites					3.6			
Chlordane isomers (sum)				1.3				
Heptachlor epoxide								
Heptachlor and heptachlor epoxide (sum)								
Dieldrin		1.8[a]	4.4	4.3		1.7		1.1
Endrin								
Aroclor 1016								
Aroclor 1221		11[a]	60	26[a]	16[a]	3.7	11[a]	2.5
Aroclor 1232		6.6[a]	29[a]	26[a]		1.9[a]		1.3[a]
Aroclor 1242			105	31	7.4	1.5	5.0	1.0
Aroclor 1248		42	4.7	26[a]	117	64	53	44
Aroclor 1254		5.7	72	280	516	2.8	353	1.9
Aroclor 1254 and 1260					99	23	50	16
Aroclor 1268			6.6	192		1.8		1.2
PCBs (sum)		119	1,190	721	516	151	353	103
Dioxins/Furans								
Dioxins/furans (TEQ)	84		2,764	3,519	13	6.1	8.9	4.2

Notes: TEQ – toxicity equivalent

[a] Ratio is halved maximum detection limit divided by the minimum screening value.

Table 5-10. Summary of COC Screening Ratios that Exceeded 1.0 for Media and Receptors In and Around Onondaga Lake

Chemical	Water	Sediments	Soil	Plants	Fish	Aquatic Wildlife[a]	Terrestrial Wildlife[b]
Metals and Cyanide							
Aluminum	•		•	•	•	•	•
Antimony		•			•	•	•
Arsenic		•	•	•	•	•	•
Barium	•		•			•	•
Beryllium			•				
Cadmium	•	•	•	•		•	•
Chromium		•	•	•	•	•	•
Cobalt						•	
Copper	•	•	•	•		•	•
Iron	•	•	•				
Lead	•	•	•	•		•	•
Manganese	•	•	•	•		•	
Mercury	•	•	•	•	•	•	•
Methylmercury	•				•	•	•
Nickel		•	•	•		•	•
Selenium			•	•	•	•	•
Silver		•	•	•			
Thallium			•	•	◘	•	•
Vanadium			•	•	•	•	•
Zinc	•	•	•	•	•	•	•
Cyanide	•		•				
Volatile Organic Compounds							
1,1,1-Trichloroethane		◘					
1,1,2,2-Tetrachloroethane		◘					
1,1,2-Trichloroethane		◘					
1,1-Dichloroethane		◘					
1,1-Dichloroethene		◘					
1,2-Dichlorobenzene		•	•			•	
1,2-Dichloroethane		◘					
1,2-Dichloroethene isomers (total)		◘					
1,3-Dichlorobenzene		•	•			•	
1,4-Dichlorobenzene		•	•			•	
Dichlorobenzenes (sum)	•	•				•	
2-Butanone		•					
2-Hexanone		◘					
4-Methyl-2-pentanone		◘					
Acetone		•					
Benzene		•	•				
Carbon disulfide		•					
Carbon tetrachloride		◘					
Chlorobenzene	•	•	•				
Chloroform		◘					
cis-1,3-Dichloropropene		◘					
trans-1,3-Dichloropropene		◘					
Ethylbenzene		•					

Table 5-10. (cont.)

Chemical	Water	Sediments	Soil	Plants	Fish	Aquatic Wildlife[a]	Terrestrial Wildlife[b]
Methylene chloride		•					
Tetrachloroethene		◘					
Toluene		•					
1,2,4-Trichlorobenzenes			•			•	•
Trichlorobenzenes (Sum)	•	•					
Trichloroethene		•					
Xylene (m,p)						•	
Xylene (o)						•	
Xylene isomers	◘	•				•	
Vinyl chloride						◘	
Semivolatile Organic Compounds							
2-Chlorophenol			◘				
1-Methylnaphthalene						•	
2-Methylnaphthalene		•				•	•
2-Methylphenol		•					
4-Bromophenyl-phenyl ether	◘	◘					
4-Chloro-3-methylphenol		◘					
4-Nitrophenol			◘				
Acenaphthene		•				•	•
Acenaphthylene						•	•
Anthracene		•	•			•	•
Benz[a]anthracene		•				•	•
Benzo[a]pyrene	◘	•	•			•	•
Benzo[g,h,i]perylene		•					•
Benzo[b]fluoranthene						•	•
Benzo[k]fluoranthene		•				•	•
Bis(2-ethylhexyl)phthalate	•				•	•	
Butylbenzyl phthalate		◘					
Chrysene		•				•	•
Dibenz[a,h]anthracene		•				•	•
Dibenzofuran		•					
Diethyl phthalate		◘					
Di-*n*-butyl phthalate		◘				◘	
Fluoranthene		•	•			•	•
Fluorene	◘	•					•
Hexachlorobenzene	◘	•	•			•	•
Hexachlorobutadiene	◘	◘				◘	◘
Hexachlorocyclopentadiene	◘	◘					
Hexachloroethane		◘				◘	◘
Indeno[1,2,3-cd]pyrene		•				•	
Naphthalene		•	•			•	•
Pentachlorophenol	◘	◘	◘			◘	◘
Phenanthrene		•	•			•	•
Phenol		•	•				
Pyrene		•	•			•	•

Table 5-10. (cont.)

Chemical	Water	Sediments	Soil	Plants	Fish	Aquatic Wildlife[a]	Terrestrial Wildlife[b]
Pesticides/Polychlorinated Biphenyls							
4,4'-DDD		•				•	
4,4'-DDE		•				•	
4,4'-DDT	◘	•				•	•
DDT and metabolites		•			•	•	
α-Chlordane	◘	•					
γ-Chlordane	◘	•					
Oxichlordane					◘		
Chlordane isomers							•
α-Endosulfan	◘	•					
β-Endosulfan	◘	◘					
Aldrin		◘	•				
Aroclor 1016		•				•	
Aroclor 1221		•				•	◘
Aroclor 1232						◘	◘
Aroclor 1242		•				•	•
Aroclor 1248		•			•	•	◘
Aroclor 1254		•			•	•	•
Aroclor 1260		•					
Aroclor 1268						•	•
PCBs (Sum)		•	•		•	•	•
β-Hexachlorocyclohexane		◘	•				
γ-Hexachlorocyclohexane		◘			•	•	
Hexachlorocyclohexanes		•					
Dieldrin		•	•			•	•
Endrin	◘	◘			•	•	
Heptachlor	◘						
Heptachlor and heptachlor epoxide (sum)		•				•	
Heptachlor epoxide	◘	◘				•	
Methoxychlor	◘	◘					
Mirex					◘		
Photomirex					◘		
Toxaphene	◘	◘					
Dioxins/Furans							
2,3,4,7,8-Pentachlorodibenzofuran					•		
2,3,7,8-Tetrachlorodibenzo-*p* -diox		•			•	•	•

Notes:

• Exceedance is based on maximum detected values divided by minimum screening values.

◘ Exceedance is based on halved maximum detection limit divided by the minimum screening value.

[a] Includes the belted kingfisher, great blue heron, osprey, mallard, tree swallow, mink, river otter, and little brown bat.

[b] Includes the short-tailed shrew and red-tailed hawk.

6. Baseline Risk Assessment Problem Formulation (ERAGS Step 3)

Step 3 of Ecological Risk Assessment Guidance for Superfund (ERAGS) initiates the problem formulation phase of the BERA (USEPA, 1997a). The components of the screening-level problem formulation are refined, taking into account various kinds of site-specific information and the concerns of stakeholders. The major components of Step 3 are as follows:

- Refinement and finalization of the list of chemicals of concern/stressors of concern (COCs/SOCs) from the list of chemicals of potential concern/stressors of potential concern (COPCs/SOPCs) identified in earlier steps.

- Further characterization of the ecological effects of the selected COCs/SOCs.

- Review of information on COC/SOC transport and fate, complete exposure pathways, and ecosystems potentially at risk.

- Refinement of assessment and measurement endpoints.

- Refinement of the conceptual site model.

These components are discussed in the following sections.

6.1 Refinement of Chemicals of Concern

The screening-level exposure estimate and risk calculations presented in Chapter 5 identified a list of preliminary COPCs for various media in Onondaga Lake. These COPCs were refined through the use of the criteria described below to derive the final list of COCs. Chemicals covered under Comprehensive Environmental Response Compensation and Liability Act of 1980 (CERCLA) Section 40 CFR Part 302.4, which lists the CERCLA hazardous substances, were considered in the COC selection. The exception to this is ammonia, which is listed as a hazardous substance in the CFR, but is treated as an SOC in this BERA since it is associated with discharges from the Metropolitan Syracuse Sewage Treatment Plant (Metro), as well as various Honeywell sites, and is a nutrient.

- **Detection Frequency.** Contaminants that were not detected in all media were dropped due to the uncertainty associated with whether they were actually present at a site and, if so, at what concentration. Frequency of detection of contaminants was a factor in deciding whether to retain them as COCs. Generally, contaminants detected in less than 5 percent of the samples were not retained, as those contaminants were considered to have limited distribution around the lake.

- **Laboratory or Field Contamination.** Infrequently detected contaminants associated with laboratory contamination or decontamination of field equipment were dropped due to the tenuous association with the site.

- **Ratios.** Ratios comparing measured COC concentrations to criteria or guidelines were calculated for water, sediment, and soil. Some media had two or more ratios representing either different sampling years or locations (for soils), all of which were considered when deciding whether to retain a contaminant.

- **Hazard Quotients.** Hazard quotients (HQs) were calculated by comparing measured tissue concentrations or modeled daily doses of chemicals to toxicity reference values (TRVs). HQs equal to or greater than 1.0 were examined closely to determine whether less conservative exposure parameters (e.g., lower bioavailability of the contaminants) could bring HQs below 1.0. Some receptors had two or more HQs, representing either different sampling years or locations, all of which were considered when deciding whether to retain a contaminant.

- **Groups of Contaminants**. Similar contaminants were grouped together to streamline COC selection and evaluation. Contaminants were individually analyzed and then summed together to calculate group exposure concentrations. Generally, these contaminants share common available TRVs and physicochemical characteristics. These groupings are generally consistent with the treatment of contaminants in the Onondaga Lake Human Health Risk Assessment (HHRA) (see Appendix A of the HHRA) (TAMS, 2002a). Metals/inorganics are not listed as a group since they are evaluated individually. Contaminants grouped together as COCs are:

  - **Polycyclic Aromatic Hydrocarbons (PAHs).** This group includes both LPAHs (low molecular weight PAHs: fluorene, naphthalene, and 2-methylnaphthalene) and HPAHs (high molecular weight PAHs: acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, fluoranthene, indeno[1,2,3-cd]pyrene, phenanthrene, and pyrene), based on the results of the principal component analysis (PCA) performed in the RI (see Appendix I of the RI report for further details [TAMS, 2002b]). Total PAHs for toxicological evaluations were calculated summing only detected values and were considered as one group due to the lack of toxicological data for most individual compounds. Distribution of PAHs in Onondaga Lake surface sediments (Chapter 8, Section 8.1.2.6) is presented for LPAHs and HPAHs.

- **DDT and Metabolites.** This group consists of 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, and other DDT metabolites.

- **Polychlorinated Biphenyls (PCBs).** This group consists of eight individual Aroclors (1016, 1221, 1232, 1242, 1248, 1254, 1260, and 1268) that were analyzed over the duration of the sampling period. The methods used for calculating total PCB concentrations are described in Chapter 8, Section 8.2.2 of this BERA.

- **Dichlorobenzenes.** This group consists of the sum of 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene.

- **Trichlorobenzenes.** This group consists of the sum of 1,2,3-trichlorobenzene, 1,2,4-trichlorobenzene, and 1,3,5-trichlorobenzene.

- **Chlordanes.** The chlordane sum consists of alpha chlordane (same as cis-chlordane), gamma chlordane (same as trans-chlordane), oxychlordane, and nonachlor (cis- and/or trans-nonachlor).

- **Heptachlor/Heptachlor Epoxide.** These two contaminants were summed and placed in one group.

- **Endosulfans.** Alpha- and beta-endosulfan were summed and placed in one group.

- **Hexachlorocyclohexanes.** Alpha-, beta-, delta-, and gamma-hexachlorocyclohexane were summed and placed in one group.

- **Dioxins and Furans.** Dioxins and furans, also known as polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) were presented in terms of toxicity equivalent (TEQ) concentrations. The TEQ approach, developed to facilitate risk assessment, generates a single toxicity value for a mixture of compounds based on the relative risk of individual constituents. Specifically, concentrations of each PCDD/PCDF congener are multiplied by their toxicity equivalence factor (TEF), which is an estimate of a PCDD/PCDF congener's toxicity relative to the most toxic congener within that chemical group (i.e., 2,3,7,8-tetrachlorodibenzo-*p*-dioxin [2,3,7,8-TCDD]), to yield compound-specific TEQ concentrations. The individual TEQ concentrations were summed, producing a single TEQ concentration that approximates the toxicity of all PCDD/PCDFs in the mixture relative to 2,3,7,8-TCDD. The TEFs used in the BERA are World Health Organization (WHO) values taken from Van den Berg et al. (1998). Sampling for PCDDs/PCDFs was performed in 2000.

Other factors considered when selecting COCs include contaminant toxicity, bioaccumulation potential, statistical distributions of contaminant concentrations (e.g., 95 percent upper confidence limits [UCLs] versus maximum detected concentrations), and USEPA guidance. Comparisons of inorganic contaminants to background concentrations was not a factor in selecting COCs, but is discussed in the uncertainty section (Chapter 11, Section 11.4, Background and Reference Concentrations) in accordance with USEPA guidance (USEPA, 2002).

6.1.1 Surface Water Chemical of Concern/Stressor of Concern Selection

A total of 32 COPCs exceeded screening values in Onondaga Lake surface water (Chapter 5, Table 5-3). Eleven contaminants, consisting of barium, copper, cyanide, lead, manganese, mercury/methylmercury, zinc, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis(2-ethylhexyl)phthalate, were retained as surface water COCs (Table 6-1).

Eighteen of these COPCs were not detected in surface water and were dropped from further consideration, as the presence of these contaminants at the lake was questionable in the absence of detected values. The undetected contaminants were: xylenes, all semivolatile organic compounds (SVOCs) except bis(2-ethylhexyl)phthalate (i.e., hexachlorobenzene, fluorene, benzo[a]pyrene, pentachlorophenol, hexachlorobutadiene, hexachlorocyclopentadiene, and 4-bromophenyl-phenyl ether), and all pesticides (i.e., alpha-chlordane, gamma-chlordane, alpha-endosulfan, beta endosulfan, endrin, heptachlor, heptachlor epoxide, methoxychlor, 4,4'-DDT, and toxaphene).

The only metals analyzed in the fall 1999 nearshore water sampling (performed mainly for HHRA purposes) were chromium, lead, manganese, mercury, and nickel. Thus, most metals were selected based on the 1992 data. Aluminum was dropped because it is biologically inactive in circumneutral to alkaline (pH 5.5 to 8.0) conditions (USEPA, 2001), and the mean pH of Onondaga Lake in 1992 was 7.7 (Appendix D, Table D-1). Iron was eliminated because it functions as a nutrient and, considering bioavailability, the ratios of 1.2 in 1992 and 2.0 in 1999 were not considered excessive. Cadmium was dropped because of its low detection frequency of 2 percent (it was detected in only 3 of 144 samples in 1992, and was not analyzed in 1999).

In addition, all SOPCs were retained for qualitative evaluation in the BERA. The SOCs consist of: ammonia, calcite, chloride, depleted dissolved oxygen (DO), nitrite, phosphorous, salinity, sulfide, and reduced water transparency. These stressors address the input of ionic waste and nutrients into the lake.

6.1.2 Sediment Chemical of Concern/Stressor of Concern Selection

The 95 contaminants that exceeded screening values in surface sediments are listed in Chapter 5, Table 5-5. A total of 30 contaminant/contaminant groups were retained as COCs based upon frequency of detection, magnitude of exceedances, and concentrations in aquatic organisms (Table 6-1). These COCs consist of:

- Thirteen inorganic contaminants: antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury/methylmercury, nickel, selenium, silver, vanadium, and zinc.

- Seven volatile organic compounds (VOCs): benzene, chlorobenzene, dichlorobenzenes (total), ethylbenzene, trichlorobenzenes (total), toluene, and xylenes.

- Four SVOCs: hexachlorobenzene, total PAHs, phenol, and dibenzofuran.

- Four pesticide groups and PCBs: chlordanes, dieldrin, heptachlor/heptachlor epoxide, DDT and metabolites, and total PCBs.

- Dioxins and furans: Total dioxins and furans.

Undetected contaminants that are not part of contaminant groups (28 of 32 COPCs) were dropped from further consideration. These were: 1,1,1-trichloroethane, 1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethane, 1,1-dichloroethene, 1,2-dichloroethene (isomers), 2-hexanone, 4-methyl-2-pentanone, carbon tetrachloride, chloroform, cis-1,3-dichloropropene, trans-1,3-dichloropropene, trichloroethene, tetrachloroethene, 4-bromophenyl-phenyl ether, 4-chloro-3-methylphenol, butylbenzyl phthalate, diethylphthalate, di-n-butyl-phthalate, hexachlorobutadiene, hexachlorocyclopentadiene, hexachloroethane, pentachlorophenol, aldrin, endrin, methoxychlor, and toxaphene.

The remaining four undetected contaminants (i.e., beta-endosulfan, beta- and gamma-hexachlorocyclohexane, heptachlor epoxide) belong to one of the groups of contaminants listed above and were examined with these groups, as discussed later in this section.

2-Methylphenol was dropped as a COC because of its low detection frequency (2 of 85 samples) in 2000 and no detections in 1992. The group endosulfans (alpha- and beta-endosulfan) was dropped from screening because beta-endosulfan was not detected in 2000 and alpha-endosulfan in 2000 had a detection rate of less than 5 percent (4 of 84 samples), and neither compound was detected in 1992 (0 of 19).

The frequency of detection (Appendix D, Tables D-5A and D-47) of the following groups were sufficient to retain them as COCs:

- PAHs (acenaphthylene, acenaphthene, anthracene, benz[a]anthracene, benzo[b]-fluoranthene, benzo[k]fluoranthene, benzo[g,h,i]perylene, benzo[a]pyrene, chrysene, dibenz[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, 1-methylnaphthalene, 2-methylnaphthalene, naphthalene, phenanthrene, and pyrene).

- DDT and metabolites (i.e., 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT).

- PCBs (i.e., Aroclors 1016, 1221, 1232, 1242, 1248, 1254, 1260, and 1268).

- Dichlorobenzenes (i.e., 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene).

- Trichlorobenzenes (i.e., 1,2,3-trichlorobenzene, 1,2,4-trichlorobenzene, and 1,3,5-trichlorobenzene).

- Chlordanes (i.e., alpha chlordane, gamma chlordane, oxychlordane, and nonachlor).

- Dioxins and furans (i.e., the sum of dioxins and furans).

The group hexachlorocyclohexanes (alpha-, beta-, delta-, and gamma-hexachlorocyclohexane) was eliminated from further consideration because individual compounds only exceeded the screening ratio for undetected values in 1992 and no individual compound had a screening ratio greater than 1.0 in 2000.

Aluminum was not retained as a sediment COC, based on draft USEPA guidance (USEPA, 2000) stating that aluminum should not be a COC at sites where the soil pH is >5.5, which applies to Onondaga Lake. Iron was eliminated as a COC because it functions as an nutrient and, assuming a bioavailability of less than 100 percent, the ratios of 1.7 in 1992 and 2.5 in 2000 were not considered excessive.

Although selenium and vanadium did not have sediment screening values (Chapter 4, Table 4-5), they were retained as COCs for fish (see Section 6.1.5) and were, therefore, also retained as COCs for sediment, as it is an exposure pathway for fish.

Acetone, methylene chloride, 2-butanone, and carbon disulfide were dropped as COCs in sediments because they may be associated with laboratory contamination or decontamination of field equipment and have no historic association with the site.

Calcite/oncolites were retained as an SOC for qualitative evaluation.

6.1.3 Wetland Surface Soils/Sediment and Dredge Spoils Area Surface Soil Chemical of Concern Selection

Wetland soils/sediments were screened against both soil and sediment guidelines and criteria (Chapter 5, Table 5-6), as many of the wetland areas are partially inundated during the year. Wetland surface soil/sediment samples were taken from 0 to 0.3 meters (m) and divided into 0 to 15 cm and 15 to 30 cm core slices. Dredge spoils surface soil samples were taken up to 107 cm in depth. Much of the dredge spoils area has been covered with fill that is believed to be from an off-site source. This fill covers the mercury-contaminated sediments dredged from the Ninemile Creek delta in the lake in the late 1960s.

Forty-one contaminants exceeded screening ratios in Onondaga Lake wetland and dredge spoils area soils/sediments (Chapter 5, Table 5-6). A total of 30 contaminants/contaminant groups were selected as soil/sediment COCs (Table 6-1). These were: antimony, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury/methylmercury, nickel, selenium, silver, thallium, vanadium, zinc, cyanide, benzene, chlorobenzene, dichlorobenzenes, trichlorobenzenes, hexachlorobenzene, phenol, total PAHs, aldrin, dieldrin, chlordanes, hexachlorocyclohexanes, DDT and metabolites, and total PCBs.

Pentachlorophenol, 2-chlorophenol, and 4-nitrophenol had ratios greater than 1.0 but were not detected in soils and were, therefore, eliminated from consideration (Chapter 5, Table 5-6).

Aluminum was not retained as a soil COC, based on draft USEPA guidance (USEPA, 2000) stating that aluminum should not be a COC at sites where the soil pH is >5.5, which applies to Onondaga Lake. Beryllium was dropped from further consideration based on an HQ of 1.1, in combination with the assumption that it was not 100 percent bioavailable from the soil.

6.1.4 Plant Chemical of Concern Selection

Only inorganic contaminant screening values (Efroymson et al., 1997a) were available for plants. A total of 14 inorganic contaminants equaled or exceeded a screening ratio of 1.0 for plants (Chapter 5, Table 5-7). Aluminum was dropped based on the draft USEPA soil guidance mentioned previously. Manganese was dropped because it has a maximum ratio of 1.0, which, in combination with lower bioavailability, results in risk below levels of concern. The remaining 12 inorganics (i.e., arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, nickel, selenium, silver, thallium, vanadium, and zinc) were the COCs selected for plant exposure (Table 6-1).

6.1.5 Fish Chemical of Concern Selection

A total of 21 contaminants exceeded screening criteria for fish (Chapter 5, Table 5-8). Eleven contaminants, consisting of antimony, arsenic, chromium, mercury/methylmercury, selenium, vanadium, zinc, endrin, total PCBs, DDT and metabolites, and dioxins/furans were selected as COCs (Table 6-1).

Photomirex, mirex, and oxychlordane were dropped because they were not detected. Thallium was dropped from consideration because it was not detected from 1992 through 1998 and had a screening ratio of less than 1.0 in 2000. Bis(2-ethylhexyl)phthalate (BEHP) was dropped because it had a ratio of 1.1 in 1992, which, in combination with lower bioavailability, results in risk below levels of concern. BEHP was not analyzed in 2000. Gamma-hexachlorocyclohexane had a ratio of 2.3 in 1992, based on one detection in 13 fish samples. In 2000, the screening ratio was 4.3×10^{-4} (Appendix D, Table D-73). Based on the initial low frequency of detection and subsequent decrease in concentration, gamma-hexachlorocyclohexane was dropped as a COC. Aluminum was dropped based on draft USEPA draft guidance (USEPA, 2000). Aroclors were grouped together in the total PCBs group.

6.1.6 Wildlife Receptor Chemical of Concern Selection

COCs for wildlife receptors were selected on a species-by-species basis using the HQ results of the screening risk assessment food-chain models (Chapter 5, Table 5-9) for the following species:

- Tree swallow (*Tachycineta bicolor*).
- Mallard (*Anas platyrhynchos*).
- Belted kingfisher (*Ceryle alcyon*).
- Great blue heron (*Ardea herodias*).
- Osprey (*Pandion haliaetus*).
- Red-tailed hawk (*Buteo jamaicensis*).
- Little brown bat (*Myotis lucifugus*).
- Short-tailed shrew (*Blarina brevida*).
- Mink (*Mustela vison*).
- River otter (*Lutra canadensis*).

Specific body weights; food, water, and sediment ingestion rates; and dietary composition were used for each receptor so that a unique group of COCs was selected for each species, despite similarities amongst some.

6.1.6.1 Tree Swallow

Twenty-one of 58 contaminants/contaminant groups with HQs equal to or greater than 1.0 were retained as COCs for a final list comprised of: arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury/methylmercury, nickel, selenium, thallium, vanadium, zinc, dichlorobenzenes, trichlorobenzenes, xylenes, bis(2-ethylhexyl)phthalate, total PAHs, DDT and metabolites, total PCBs, and dioxins/furans (Table 6-2).

Vinyl chloride, pentachlorophenol, di-n-butyl phthalate, hexachloroethane, and hexachlorobutadiene were dropped as COCs because they were undetected in sediment, which was used to model aquatic invertebrate concentrations, or water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.2 Mallard

Fifteen of 45 contaminants/contaminant groups with HQs equal to or greater than 1.0 were retained as COCs for a final list comprised of: barium, cadmium, chromium, cobalt, copper, mercury/methylmercury, nickel, vanadium, zinc, dichlorobenzenes, trichlorobenzenes, xylenes, total PAHs, total PCBs, and dioxins/furans (Table 6-2). Thallium and 4,4'-DDT were dropped because they had ratios of 1.1, and with alternative assumptions (e.g., mean weight, ingestion rate, bioavailability) HQs would fall below 1.0.

Vinyl chloride, pentachlorophenol, di-n-butyl phthalate, hexachloroethane, and hexachlorobutadiene were dropped as COCs for the mallard because they were undetected in sediment and water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.3 Belted Kingfisher

Eleven of 26 contaminants/contaminant groups with HQs equal to or greater than 1.0 were retained as COCs for a final list comprised of: chromium, lead, mercury/methylmercury, selenium, zinc, total PAHs, hexachlorocyclohexanes, DDT and metabolites, endrin, total PCBs, and dioxins/furans (Table 6-2).

Pentachlorophenol and hexachloroethane were dropped as COCs for the belted kingfisher because they were undetected in fish, sediment, or water. Bis(2-ethylhexyl)phthalate and heptachlor/heptachlor epoxide were dropped due to ratios of 1.0 and 1.1, respectively, in 1992 to 1998 and HQs below 1.0 in the 1999 to 2000 sampling, indicating that concentrations of these two contaminants have decreased below risk levels. The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.4 Great Blue Heron

Eight of 19 contaminants/contaminant groups with HQs equal to or greater than 1.0 were retained as COCs for a final list comprised of: chromium, mercury/methylmercury, selenium, zinc, total PAHs, hexachlorocyclohexanes, DDT and metabolites, and total PCBs (Table 6-2).

Pentachlorophenol and hexachloroethane were dropped as COCs for the great blue heron because they were undetected in fish, sediment, or water. The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.5 Osprey

Eight of 18 contaminants/contaminant groups with HQs above 1.0 were retained as COCs for a final list comprised of: chromium, mercury/methylmercury, selenium, zinc, hexachlorocyclohexanes, DDT and metabolites, total PCBs, and dioxins/furans (Table 6-2).

Endrin was eliminated based on a HQ of 1.0 because, with alternative assumptions (e.g., mean weight, ingestion rate, bioavailability), the HQs would fall below 1.0. The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.6 Red-Tailed Hawk

Six of 24 contaminants/contaminant groups were retained as COCs for a final list comprised of chromium, lead, mercury/methylmercury, total PAHs, DDT and metabolites, and dioxins/furans (Table 6-2).

Pentachlorophenol and hexachloroethane were dropped as COCs for the red-tailed hawk because they were undetected in soil or water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.7 Little Brown Bat

Twenty-two of 52 contaminants/contaminant groups were retained as COCs for a final list comprised of: antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury/methylmercury, nickel, selenium, thallium, vanadium, zinc, trichlorobenzenes, xylenes, total PAHs, hexachlorobenzene, total PCBs, dieldrin, and dioxins/furans (Table 6-2).

Vinyl chloride, pentachlorophenol, hexachloroethane, and hexachlorobutadiene were dropped as COCs because they were not detected in sediment (used to model aquatic invertebrate concentrations) or water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.8 Short-Tailed Shrew

Eighteen of 47 contaminants/contaminant groups were retained as COCs for a final list comprised of: arsenic, barium, cadmium, chromium, lead, mercury/methylmercury, nickel, selenium, thallium, vanadium, zinc, trichlorobenzenes, total PAHs, hexachlorobenzene, chlordanes, dieldrin, total PCBs, and dioxins/furans (Table 6-2).

Pentachlorophenol, hexachloroethane, and hexachlorobutadiene were dropped as COCs for the short-tailed shrew because they were not detected in soil or water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). Copper and nickel were dropped based on HQs of 1.1 for both these elements, which would likely go below 1.0 if mean body weights and food intake assumptions were used, or if lower bioavailability was assumed. The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.9 Mink

Eleven of 26 contaminants/contaminant groups were retained as COCs for a final list comprised of: arsenic, chromium, mercury/methylmercury, selenium, vanadium, total PAHs, DDT and metabolites, dieldrin, hexachlorobenzene, total PCBs, and dioxins/furans (Table 6-2).

Pentachlorophenol and hexachloroethane were dropped as COCs for the mink because they were undetected in fish, sediment, and water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). Antimony was dropped because of the low overall detection rate in fish (2 of 4 samples in 1992 and 0 of 55 samples in 2000), which drives the mink food-web model. Thallium was dropped as a COC because, although the HQ was 3.1 from 1992 to 1998 and 1.1 in 1999 to 2000, it was not detected in fish in 1992 and was only detected in one of 55 fish analyzed in 2000. The remaining contaminants were evaluated separately or in groups as COCs.

6.1.6.10 River Otter

Nine of 23 contaminants/contaminant groups were retained as COCs for a final list comprised of: arsenic, chromium, mercury/methylmercury, selenium, vanadium, total PAHs, DDT and metabolites, total PCBs, and dioxins/furans (Table 6-2).

Pentachlorophenol and hexachloroethane were dropped as COCs for the river otter because they were undetected in fish, sediment, and water. Aluminum was dropped based on draft USEPA guidance (USEPA, 2000). Antimony was dropped because of the low overall detection rate in fish (2 of 4 samples in 1992 and 0 of 55 samples in 2000), which drives the river otter food-web model. Thallium was dropped as a COC because although the HQ was 2.1 in 1992, thallium was not detected in fish in 2000. Dieldrin was eliminated with a HQ of 1.1, since with alternative assumptions (e.g., mean weight, ingestion rate, bioavailability) the HQ would fall below 1.0. The remaining contaminants were evaluated separately or in groups as COCs.

6.2 Further Characterization of Ecological Effects

Screening-level effect levels were selected and addressed in Chapter 4, Section 4.2. A review of previously identified literature and new literature searches were performed to further characterize selected COCs. The Ovid search engine was used to retrieve abstracts on the toxicity of selected COCs to vertebrate receptors (i.e., fish, amphibians, reptiles, birds, and mammals) and the life-history characteristics of receptors. To assist in the selection of toxicity values (i.e., no observed adverse effect levels [NOAELs] and lowest observed adverse effect levels [LOAELs]) and receptor parameters, abstracts were reviewed and original papers selected from the searches were obtained. The TRVs selected for use in the BERA are discussed in Chapter 9, Section 9.3, and the life history characteristics of receptors are covered in Chapter 8, Section 8.2.

6.3 Contaminant Transport and Fate, Ecosystems Potentially at Risk, and Complete Exposure Pathways

6.3.1 Contaminant Transport and Fate

Contaminant transport and fate are a function of the physical and chemical characteristics of the contaminant, as well as the system through which it may be transported. An important chemical characteristic for contaminants in aquatic systems is solubility in water. The Onondaga Lake COCs include both water-soluble (e.g., manganese, nickel) and relatively insoluble (e.g., PCBs, dioxin/furans) contaminants. Water-soluble contaminants are transported primarily in dissolved form in surface water and tend to remain in solution, potentially exiting the lake at the outlet. Volatilization can also affect the transport and fate of volatile COCs.

The transport and fate of relatively insoluble contaminants parallels that of particles (especially particulate organic carbon, in the case of most organic contaminants). These insoluble contaminants can be carried short distances on particles before settling to the sediment bed. Sediment is continuously deposited in depositional regions of the lake, resulting in profiles of various levels of contamination at different locations. Buried contaminants may be exposed by processes such as bioturbation and scour. Contaminant deposits may also be resuspended and transported by waves and currents to locations within the lake and connected wetlands and outside of the lake, via the lake outlet.

6.3.1.1 Mercury Methylation

One of the key contaminants present in Onondaga Lake is mercury, which is of concern because inorganic and organic forms of mercury can be converted into the highly toxic methylmercury. Parts of the following discussion of methylation and bioaccumulation of mercury were taken from the National Oceanic and Atmospheric Administration's (NOAA's) report on mercury (NOAA, 1996).

Mercury Methylation in Sediments

Methylation in aquatic habitats is primarily a biological process. Mono- and dimethylmercury are formed by microorganisms in both sediment and water through the methylation of inorganic mercuric ions (Hg[II]). Dimethylmercury, which is highly volatile, is generally not persistent in aquatic environments. Methylation is influenced by environmental variables that affect both the availability of mercuric ions for methylation and the growth of the methylating microbial populations. Methylation rates are higher under anoxic conditions, in freshwater compared to saltwater, and in low-pH environments. The presence of organic matter can stimulate growth of microbial populations (and reduce oxygen levels), thereby enhancing the formation of methylmercury. Sulfide can bind mercury and limit methylation.

Methylmercury production can vary due to seasonal changes in nutrients, oxygen, temperature, and hydrodynamics. In most studies, methylation increased during the summer months when biological productivity was high, and decreased during the winter months. Measurements of total mercury concentrations in sediment do not provide information on the form of mercury present, methylation potential, or availability to organisms locally and downstream. If environmental conditions are conducive for methylation, methylmercury concentrations may be high, as compared to the supply and distribution of total mercury.

Mercury Methylation in Wetlands

Mercury methylation has been reported to occur in wetlands. As measured by the US Geological Survey (USGS), methylmercury comprises about 1 to 10 percent of total mercury in sediments of aquatic ecosystems (e.g., from streams and/or wetlands sediments in mixed agricultural/forest areas, abandoned mines, urban areas, etc.), in the US (Krabbenhoft et al., 1999). Krabbenhoft et al. (1999) found that methylmercury production was proportional to total mercury concentrations at low sediment concentrations, but at high concentrations (>1 parts per million [ppm]), little additional methylmercury was produced with increasing mercury. Sediments in mining and urban areas were found to have the lowest methylation efficiency.

Gilmour et al. (1998) studied mercury methylation in Florida Everglades wetlands. Methylation rates averaged between about 0.1 and 2 percent. The highest rates were seen in southern wetlands with lower nutrient concentrations, sulfate, and sulfide concentrations, which also had higher total mercury concentrations (up to about 0.4 ppm). The increase in methylmercury was considered to be driven by factors other than total mercury, because methylmercury concentrations increased by a factor of about 25, while total mercury increased only by a factor of 3 to 4.

In sediment samples collected by Honeywell in the West Flume, ditches, and ponded areas/wetlands at the LCP Bridge Street site in 1995 and 1996 (see Appendix G for site summary), methylmercury comprised between 0.003 and 2.2 percent of the total mercury found, with an average of 0.25 percent (Table 6-3). The average total mercury concentration was 32 mg/kg (ppm). The highest proportion of methylmercury was generally seen in samples with lower concentrations of total mercury (e.g., 3 mg/kg or less), confirming Krabbenhoft et al.'s observations (1999).

Onondaga Lake is an eutrophic system with high sulfide concentrations (sulfide inhibits methylmercury production), and is likely to have a wetland mercury methylation rate of 1 percent or less, similar to the eutrophic sites studied in the Florida Everglades. Average mercury concentrations for Wetlands SYW-6 and SYW-12 were 1.3 and 0.7 mg/kg, respectively (Appendix H, Tables H-17 and H-19). If total mercury concentrations are a main driving factor, these Onondaga Lake site wetlands are likely to have mercury methylation rates at the upper end of their expected range.

Based on the literature and LCP Bridge Street site data, a wetland mercury methylation rate of 1 percent is considered to be protective of the Onondaga Lake ecosystem for use in this BERA. No mercury methylation is assumed to occur in the dredge spoils area.

Mercury Methylation in Biota

Mercury is accumulated by fish, invertebrates, mammals, and aquatic plants, and its concentration tends to increase with increasing trophic level. Although inorganic mercury is the dominant form of mercury in the environment and is easily taken up, it is also depurated relatively quickly.

Methylmercury accumulates quickly, depurates very slowly, and, therefore, biomagnifies in higher trophic species. The percentage of methylmercury, as compared to total mercury, also increases with age in both fish and invertebrates. Uptake and depuration rates vary between tissues within an organism. Partitioning of mercury between tissues within aquatic organisms is influenced by the chemical form of mercury and route of exposure (ingestion or via the gills). Due to its preferential uptake, ability to be transferred among tissues, and slow depuration, most of the mercury (ranging between 80 to 99 percent [Huckabee et al., 1979; Chvojka, 1988; Grieb et al., 1990; Southworth et al.,1995]) in fish muscle tissue is methylmercury. NYSDEC Onondaga Lake fish samples from 1992 that were analyzed for both mercury and methylmercury indicated that mercury and methylmercury data are essentially interchangeable; that is, the methylmercury result was generally within 5 percent of the total mercury result. Based on the 1992 results, only mercury was analyzed in the 2000 fish sampling, and all of it was assumed to be methylmercury.

While sediment is usually the primary source of mercury in most aquatic systems, the food web is the main pathway for accumulation. High trophic level species tend to accumulate the highest concentrations of mercury, with the greatest concentrations in fish-eating predators. Methylmercury accumulates in aquatic food chains in which the top-level predators usually contain the highest concentrations. Correlations have been made between sediment and lower trophic species that typically have a high percentage of inorganic mercury, and between mercury concentrations in higher trophic species and their prey items. The best measure of bioavailability of mercury in any system is obtained by analyzing mercury concentrations in the biota at the specific site. Concentrations of methylmercury and other contaminants in fish and upper trophic level organisms can remain high after concentrations have decreased in sediment and water, due to the slow rate of depuration of methylmercury from fish tissues (e.g., Eisler, 1987a; Wiener and Spry, 1996).

6.3.1.2 Organic Compounds

Biodegradation of organic contaminants can be significant for certain contaminants under conditions favoring bacterial activity. However, most organic COCs in Onondaga Lake are relatively recalcitrant, with half-lives extending into years, especially under the anoxic conditions (Howard et al., 1991; Mackay et al., 1992) that are expected in deeper sediment. Contaminant transport and fate of COCs in Onondaga Lake is discussed in greater detail in the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

6.3.2 Ecosystems Potentially at Risk

Ecosystems potentially at risk include those associated with the surface water, sediments, and bordering wetlands and terrestrial areas of Onondaga Lake. Descriptions of the aquatic environment and terrestrial habitats and the species found in them are provided in Chapter 3, Sections 3.2.4 and 3.2.5. Within these ecosystems, aquatic organisms (e.g., plankton, benthic macroinvertebrates, and fish), semiaquatic organisms (e.g., amphibians, some reptiles, some birds and mammals), terrestrial organisms (e.g., some reptiles, most birds and mammals), and plants are potentially at risk from exposure to COCs in water, sediment, soil, and prey. Animals feeding on prey from the lake can be exposed to elevated concentrations of chemicals due to the bioaccumulation potential of some of the contaminants (e.g., mercury, PCBs) present in the lake.

COCs can impact the lake ecosystem at the organism, population, and community levels. For example, ecological risk to benthic macroinvertebrates and fish can manifest itself as adverse impacts on reproduction and growth of individual organisms, abundance and distribution of populations, or community structure. For wildlife species, risk can manifest itself in diverse ways such as adverse impacts on organism growth, reproduction, behavior, and cellular/organ functions. The effects of some contaminants, particularly those affecting endocrine functions, may not show up until one or two generations after exposure.

6.3.3 Complete Exposure Pathways

Complete exposure pathways via direct contact/ingestion and bioaccumulation exist for organisms associated with surface water, sediment, and soil in and around Onondaga Lake. Direct contact with and ingestion of surface water, sediments, and prey (e.g., zooplankton, benthic invertebrates, eggs, and small fish) can expose aquatic animals, such as benthic macroinvertebrates and fish, to COCs. Exposure to contaminated lake water during sensitive development times of aquatic eggs and embryos can affect the viability of some organisms breeding in the lake. Direct contact with surface water is only discussed qualitatively in this assessment, due to limited exposure data. Reptiles and amphibians are also exposed to COCs via direct contact with and ingestion of surface water, sediments, soils, and prey.

Terrestrial species such as birds and mammals can be exposed to COCs through direct contact with and/or ingestion of surface water, sediments, soil, and prey (aquatic, semiaquatic, and terrestrial organisms). Wetland and terrestrial plants can be exposed to COCs through direct contact with surface water and uptake of contaminants from sediments and soils.

Bioaccumulation at each level of the food web can increase the contaminant exposure concentration to many times the original concentration found in water, sediments, and soil. A complete exposure pathway via bioaccumulation exists for upper trophic level species (e.g., insectivorous, piscivorous, and carnivorous fish, birds, and mammals) for COCs that bioaccumulate, such as methylmercury and PCBs.

6.4 Selection of Assessment Endpoints

USEPA guidance states, "Superfund risk assessment should use site-specific assessment endpoints that address chemical-specific potential adverse effects to local populations and communities of plants and animals" (USEPA, 1999a). Consistent with this guidance, assessment endpoints for this BERA were selected, taking into account their biological significance, their susceptibility to potential contact through indirect or direct exposure to COCs, the availability of pertinent assessment models, and toxicological information in the literature. Risks to individual fish and wildlife receptors are used to assess risks to these populations. The assessment endpoints selected during screening (Chapter 4, Section 4.1.3.1) were retained for the BERA, as follows:

- Sustainability (i.e., survival, growth, and reproduction) of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife.

- Sustainability (i.e., survival, growth, and reproduction) of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife.

- Sustainability (i.e., survival, growth, and reproduction) of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife.

- Sustainability (i.e., survival, growth, and reproduction) of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife.

- Sustainability (i.e., survival, growth, and reproduction) of a benthic invertebrate community that can serve as a food source for local fish and wildlife.

- Sustainability (i.e., survival, growth, and reproduction) of local fish populations.

- Sustainability (i.e., survival, growth, and reproduction) of local amphibian and reptile populations.

- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local benthivorous waterfowl populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local carnivorous bird populations.

- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous mammal populations.

- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous mammal populations.

6.5 Selection of Measurement Endpoints and Associated Risk Questions

Measurement endpoints provide the actual values used to evaluate attainment of each assessment endpoint. For the Onondaga Lake BERA, the measurement endpoints (in relation to their respective assessment endpoints) are phrased as in relation to respective risk questions, as follows:

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife.

Does the macrophyte community structure reflect the influence of COCs/SOCs?

Measurement Endpoint 1: Field observations of the abundance, distribution, and species composition of local macrophyte communities in relation to COCs/SOCs in water and sediments and habitat characteristics.

Do the contaminants/stressors present in Onondaga Lake sediment affect macrophyte growth and survival?

Measurement Endpoint 2: Greenhouse studies of macrophyte growth and survival on field-collected sediments and macrophyte transplant studies in Onondaga Lake.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 3: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared to state and federal water quality values and qualitative evaluation of narrative standards.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife.

Does the phytoplankton community structure reflect the influence of COCs/SOCs?

Measurement Endpoint 1: Field observations of the abundance and species composition of local phytoplankton communities in relation to COCs/SOCs in water and sediments and habitat characteristics.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared with state and federal water quality values and qualitative evaluation of narrative standards.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife.

Does the zooplankton community structure reflect the influence of COCs/SOCs?

Measurement Endpoint 1: Field observations of the historical abundance and species composition of local zooplankton communities in relation to COCs/SOCs in water and sediments and habitat characteristics and studies of zooplankton hatching success.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared with state and federal water quality values and qualitative evaluation of narrative standards.

Do measured concentrations of contaminants and stressors in sediments exceed criteria and/or guidelines for the protection of aquatic organisms?

Measurement Endpoint 3: Measured average and 95 percent UCL concentrations of COCs/SOCs in sediments compared to state and federal sediment quality values.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife.

Does the terrestrial plant community structure reflect the influence of COCs/SOCs?

Measurement Endpoint 1: Field observations of the abundance and species composition of local plant communities in relation to COCs/SOCs in soils and habitat characteristics.

Do measured concentrations of contaminants and stressors in soil exceed toxicity values for terrestrial plants?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs/SOCs in soil compared with literature plant toxicity values.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of a benthic invertebrate community that can serve as a food source for local fish and wildlife.

Does the benthic community structure reflect the influence of COCs/SOCs?

Measurement Endpoint 1: Field observations of the abundance and species composition of local benthic macroinvertebrate communities in relation to COCs/SOCs in water and sediments and habitat characteristics using benthic metrics.

Do concentrations of contaminants and stressors in sediment influence mortality, growth, or fecundity of invertebrates living in or on lake sediments?

Measurement Endpoint 2: Sediment toxicity based on laboratory tests of field-collected sediments using sensitive and representative benthic macroinvertebrate species and a variety of test endpoints.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 3: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared to state and federal water quality values and qualitative evaluation of narrative standards.

Do measured concentrations of contaminants and stressors in sediment exceed levels that may adversely affect benthic invertebrates and/or criteria and/or guidelines for the protection of aquatic organisms?

Measurement Endpoint 4: Measured concentrations of COCs in sediment compared to site-specific sediment effects concentrations (SECs) and consensus probable effect concentrations (PECs) and measured average and 95 percent UCL concentrations of COCs/SOCs in sediments compared to state and federal sediment quality values.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local fish populations.

What does the fish community structure suggest about the health of local fish populations?

Measurement Endpoint 1: Field observations of the abundance, distribution, and species composition of local fish communities in relation to COCs/SOCs in water and sediments and habitat characteristics as compared to those in similar lakes in New York State.

Has the presence of contaminants and/or stressors influenced fish foraging or nesting activities?

Measurement Endpoint 2: Field observations of suitable nesting habitat and populations of juveniles in relation to COCs/SOCs and habitat characteristics.

Do fish found in Onondaga Lake show reduced growth or increased incidence of disease (e.g., tumors) as compared to fish from other lakes?

Measurement Endpoint 3: Observations of disease as compared to those in New York reference lakes.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 4: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared to state and federal water quality values and qualitative evaluation of narrative standards.

Do measured concentrations of contaminants and stressors in sediments exceed criteria and/or guidelines for the protection of aquatic organisms (applicable to benthic-dwelling fish)?

Measurement Endpoint 5: Measured average and 95 percent UCL concentrations of COCs/SOCs in sediments compared to state and federal sediment quality values.

Do measured concentrations of contaminants in fish exceed TRVs for adverse effects on fish mortality or reproduction?

Measurement Endpoint 6: Measured average and 95 percent UCL COC concentrations in fish compared to TRVs.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local amphibian and reptile populations.

What do the available field-based observations suggest about the health of local amphibian and reptile communities?

Measurement Endpoint 1: Field observations of the abundance and species composition of local communities of amphibians and reptiles in relation to COCs/SOCs in water, sediments, and soils and habitat characteristics.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of aquatic organisms?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs/SOCs in water compared to state and federal water quality values and qualitative evaluation of narrative standards.

Have laboratory studies indicated the potential for adverse effects to amphibian embryos from exposure to Onondaga Lake water?

Measurement Endpoint 3: Results of amphibian embryos exposed to unfiltered lake water as compared to filtered water or controls.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local insectivorous bird populations.

Do modeled dietary doses to insectivorous birds exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC concentration dietary doses based on measured and modeled concentrations of COCs in lake media (i.e., surface water and invertebrates), compared with TRVs.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of wildlife?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs in surface water compared to state and federal water quality criteria for protection of wildlife.

What do the available field-based observations suggest about the health of local insectivorous bird populations?

Measurement Endpoint 3: Field observations of insectivorous birds around Onondaga Lake.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local benthivorous waterfowl populations.

Do modeled dietary doses to benthivorous waterfowl exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC concentrations based on measured and modeled concentrations of COCs (i.e., surface water, sediment, and invertebrates) in lake media compared with TRVs.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of wildlife?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs in surface water compared to state and federal water quality criteria for protection of wildlife.

What do the available field-based observations suggest about the health of local waterfowl populations?

Measurement Endpoint 3: Field observations of waterfowl around Onondaga Lake.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local piscivorous bird populations.

Do modeled dietary doses to piscivorous birds exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC dietary doses based on measured concentrations of COCs in lake media (i.e., surface water, sediment, and fish), compared with TRVs.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of wildlife?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs in surface water compared to state and federal water quality criteria for protection of wildlife.

What do the available field-based observations suggest about the health of local piscivorous bird populations?

Measurement Endpoint 3: Field observations of piscivorous birds around Onondaga Lake.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local carnivorous bird populations.

Do modeled dietary doses to carnivorous birds exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC dietary doses based on measured concentrations of COCs in media (i.e., surface water, soil, and small mammals), compared with TRVs.

What do the available field-based observations suggest about the health of local carnivorous bird populations?

Measurement Endpoint 2: Field observations of carnivorous birds around Onondaga Lake.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local insectivorous mammal populations.

Do modeled dietary doses to insectivorous mammals exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC dietary doses based on measured and modeled concentrations of COCs in lake media (i.e., surface water, soil, and invertebrates), compared with TRVs.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of wildlife?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs in surface water compared to state and federal water quality criteria for protection of wildlife.

What do the available field-based observations suggest about the health of local insectivorous mammal populations?

Measurement Endpoint 3: Field observations of insectivorous mammals around Onondaga Lake.

Assessment Endpoint: Sustainability (i.e., survival, growth, and reproduction) of local piscivorous mammal populations.

Do modeled dietary doses to piscivorous mammals exceed TRVs for adverse effects on reproduction?

Measurement Endpoint 1: Modeled average and 95 percent UCL COC dietary doses based on measured concentrations of COCs in lake media (i.e., surface water, sediment, soil, and fish), compared with TRVs.

Do measured concentrations of contaminants and stressors in surface water exceed standards, criteria, and guidance for the protection of wildlife?

Measurement Endpoint 2: Measured average and 95 percent UCL concentrations of COCs in surface water compared to state and federal water quality criteria for protection of wildlife.

What do the available field-based observations suggest about the health of local piscivorous mammal populations?

Measurement Endpoint 3: Field observations of piscivorous mammals around Onondaga Lake.

Given the limitations of the available data, some of the selected measurement endpoints do not provide direct measures of the assessment endpoints. In such cases, every effort has been made to evaluate the implications of the measurement endpoint results for the assessment endpoints, and the resulting uncertainties are acknowledged and discussed further in Chapter 11, Uncertainty Analysis.

6.6 Conceptual Model

The preliminary conceptual model for Onondaga Lake was presented in Chapter 4, Section 4.1 (Figure 4-1) and remains unchanged for this ERAGS step. The major potential sources of contaminants to the lake are point-source discharges, tributaries, and groundwater. From these potential sources, contaminants can enter lake water through inflow, can enter sediments through precipitation and deposition, and can enter biota through direct contact, respiration, and ingestion. Contaminants can also enter lake water from resuspension of the in-lake waste deposit and contaminated sediments. Therefore, potential secondary sources are the water, sediments, and biota of the lake. Potentially toxic chemicals in secondary sources can result in exposure to aquatic and semiaquatic ecological receptors through direct contact, respiration, and ingestion. These chemicals can also reach terrestrial receptors through direct contact and ingestion. Additional potential stressors include ionic waste (calcium, chloride, and sodium), nutrients (i.e., nitrite, phosphorus, sulfide), calcite deposits (including oncolites), salinity, ammonia, depleted DO, and reduced water transparency.

Table 6-1. Contaminants of Concern Selected for Onondaga Lake Media

Chemical	Water	Sediment	Soil	Plants	Fish
Metals					
Antimony		•	•		•
Arsenic		•	•	•	•
Barium	•		•		
Cadmium		•	•	•	
Chromium		•	•	•	•
Copper	•	•	•	•	
Iron			•		
Lead	•	•	•	•	
Manganese	•	•	•		
Mercury/Methylmercury	•	•	•	•	•
Nickel		•	•	•	
Selenium		•	•	•	•
Silver		•	•	•	
Thallium			•	•	
Vanadium		•	•	•	•
Zinc	•	•	•	•	•
Cyanide	•		•		
Volatile Organic Compounds					
Benzene		•	•		
Chlorobenzene	•	•	•		
Dichlorobenzenes (Sum)	•	•	•		
Ethylbenzene		•			
Toluene		•			
Trichlorobenzenes (Sum)	•	•	•		
Xylene isomers		•			
Semivolatile Organic Compounds					
Bis(2-ethylhexyl)phthalate	•				
Dibenzofuran		•			
Hexachlorobenzene		•	•		
Phenol		•	•		
Polycyclic aromatic hydrocarbon (total)		•	•		
Pesticides/Polychlorinated Biphenyls					
Aldrin			•		
Chlordane isomers		•	•		
DDT and metabolites		•	•		•
Dieldrin		•	•		
Endrin					•
Hexachlorocyclohexanes			•		
Heptachlor and heptachlor epoxide		•			
Polychlorinated biphenyls (total)		•	•		•
Dioxins/Furans					
Total dioxins/furans		•			•

Note: • – Contaminants of concern assessed in the BERA for the specific media listed.
DDT – dichlorodiphenyltrichloroethane

Table 6-2. Contaminants of Concern for Wildlife Species Evaluated for the Onondaga Lake BERA

Contaminants of Concern	Tree Swallow	Mallard	Belted Kingfisher	Great Blue Heron	Osprey	Red-Tailed Hawk	Little Brown Bat	Short-Tailed Shrew	Mink	River Otter
Metals										
Antimony							•	•		
Arsenic	•						•	•	•	•
Barium	•	•					•	•		
Cadmium	•	•					•	•		
Chromium	•	•	•	•	•	•	•	•	•	•
Cobalt	•	•					•			
Copper	•	•					•			
Lead	•		•			•	•	•		
Manganese							•			
Mercury/Methylmercury	•	•	•	•	•	•	•	•	•	•
Nickel	•	•					•			
Selenium	•		•	•	•		•	•	•	•
Thallium	•						•	•		
Vanadium	•	•					•	•	•	•
Zinc	•	•	•	•	•		•	•		
Volatile Organic Compounds										
Dichlorobenzenes (total)	•	•								
Trichlorobenzenes (total)	•	•					•	•		
Xylenes (total)	•	•					•			
Semivolatile Organic Compounds										
Bis(2-ethylhexyl)phthalate	•									
Hexachlorobenzene							•	•	•	
Polycyclic aromatic hydrocarbon (total)	•	•	•	•		•	•	•	•	•
Pesticides/Polychlorinated Biphenyls										
Chlordanes								•		
DDT and metabolites	•		•	•	•	•			•	
Dieldrin							•	•	•	•
Endrin			•							
Hexachlorocyclohexanes			•	•	•					
Polychlorinated biphenyls (total)	•	•	•	•	•		•	•	•	•
Dioxins/Furans										
Dioxins/furans (TEQ)	•	•	•		•	•	•	•	•	•

Notes:

• – Contaminants of concern assessed in the BERA for the specific receptor listed.

DDT – dichlorodiphenyltrichloroethane

TEQ – toxicity equivalent

Table 6-3. Mercury:Methylmercury Ratios in Samples Collected at the LCP Bridge Street Site

Depth (cm)	Sample Location	Log Notes (location characteristics)	Total Mercury (mg/kg dw)	Methyl-mercury (µg/kg dw)	MeHg/Hg Percent
0 - 6	West Flume to north of property boundary	marsh vegetated	23.0 J	11.0 J	0.05%
0 - 6	West Flume to east of property boundary	flume	0.6	1.2 J	0.21%
0 - 6	West Flume at Geddes Brook	marsh vegetated	28.6 J	7.8 J	0.03%
0 - 6	Mouth of west ditch at ponded area	veg. ditch approx 1 in. deep	35.8	13.3 J	0.04%
0 - 6	Mouth of east ditch at West Flume	veg. ditch approx 3-4 in deep	4.4	5.2	0.12%
0 - 6	East ditch	grassy ditch	24.2 J	15.9 J	0.07%
0 - 6	East ditch	ditch in marsh	1.8	3.8	0.21%
0 - 6	Ponded area	unvegetated ditch	51.5	13.2	0.03%
0 - 6	Ponded area	unvegetated ditch	131.0 J	14.6	0.01%
0 - 6	Ponded area	vegetated ditch	10.2 J	11.5 J	0.11%
0 - 6	On-site drainage ditch to east of west ditch	approx 4 - 6 in deep	57.7 J	26.3	0.05%
0 - 6	On-site drainage ditch to east of west ditch	approx 8 in deep	193.0 J	175.0 J	0.09%
0 - 6	West ditch	veg. ditch less than 1 in deep	29.8	15.8 J	0.05%
0 - 6	Ponded area at West Flume	veg. marsh approx 2 in deep	56.0	3.6 J	0.01%
0 - 6	Ditch by west plant wall	lined ditch	2.9	63.8	2.20%
0 - 15	Ponded area, by west property boundary	vegetated, approx 0.5 ft deep	56.3 J	14.0 J	0.02%
0 - 15	Ponded area, by west property boundary	center of area, approx 4 in deep	56.4 J	11.4 J	0.02%
0 - 15	Ponded area	vegetated 4 in deep	9.3 J	3.8	0.04%
0 - 15	East ditch	vegetated, approx 3 in deep	9.5 J	29.7	0.31%
0 - 15	Ponded area	vegetated, approx 4 in deep	21.5 J	19.4	0.09%
0 - 15	Ponded area	vegetated, approx 3 in deep	41.9 J	1.2	0.003%
0 - 15	Ponded area	vegetated, approx 4 in deep	7.5 J	2.4	0.03%
0 - 15	Ponded area	vegetated, very moist, no water	12.6 J	74.3	0.59%
0 - 15	Ponded area	vegetated moist area, no water	1.8 J	20.1	1.12%
0 - 15	Ponded area	vegetated moist area, no water	1.5 J	6.2	0.41%
0 - 15	Ponded area	vegetated moist area, no water	1.7 J	12.2	0.72%
0 - 15	West Flume west of property boundary	vegetated, approx 2 in deep	11.5 J	14.6 J	0.13%
0 - 15	West Flume west of property boundary	vegetated moist area, no water	18.5 J	31.7 J	0.17%
0 - 15	West Flume west of property boundary	vegetated moist area, no water	23.3 J	68.9 J	0.30%
		Max. Conc./Ratio	193	74	2.20%
		Min. Conc./Ratio	0.6	1.2	0.003%
		Ave. Conc./Ratio	32	25	0.25%

Notes:
1. J indicates an estimated value.
2. Sampling conducted by Parsons Engineering Science, Inc.

7. STUDY DESIGN (ERAGS STEPS 4 AND 5)

As discussed in Chapter 1, major field investigations were conducted by Honeywell in 1992, 1999, and 2000 to provide information for the BERA. Additional fish data collected by NYSDEC from 1992 through 2000 and wetland data from 2002 were also used in this BERA. The major components of each investigation are described below, and are discussed in detail in Chapter 2 of the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

7.1 1992 Investigation

The 1992 field investigation, conducted from April to December of 1992 by Honeywell/PTI, was subdivided into five smaller investigations corresponding to the major types of data targeted for collection. These smaller investigations are described below, along with information from each investigation that was used in the BERA:

- **Geophysical Investigation** – Information on the bathymetry of Onondaga Lake was used to stratify benthic macroinvertebrate sampling stations by water depth and to evaluate the potential for wind-induced sediment disturbance throughout the littoral zone of the lake.

- **Contaminant and Stressor Investigation** – Information on contaminants and stressor concentrations and distribution in surface sediments (0 to 2 cm) of Onondaga Lake was used to evaluate potential risks to biota in the lake.

- **Mercury and Calcite Mass Balance Investigation** – Information on mercury and calcite concentrations in the water of Onondaga Lake and its tributaries was collected. However, Honeywell did not develop acceptable mass balance models for use in the BERA (NYSDEC/TAMS, 1998b,c). Mass balance estimates for mercury prepared by NYSDEC/TAMS are included in the RI report (TAMS, 2002b).

- **Ecological Effects Investigation** – Quantitative information on sediment chemistry, toxicity, and benthic macroinvertebrate communities in Onondaga Lake, as compared to a nearby reference lake (i.e., Otisco Lake), was used to evaluate potential risks to sediment-dwelling organisms in Onondaga Lake. Semi-quantitative and qualitative information on macrophyte, phytoplankton, and zooplankton communities in Onondaga Lake was combined with more quantitative information collected by other parties to evaluate potential risks to those communities in the lake.

- **Bioaccumulation Investigation** – Information on chemical of concern (COC) concentrations in sediment, surface water, benthic macroinvertebrates, and fish in Onondaga Lake was used to evaluate exposure to COCs and potential risks to fish, semiaquatic, and terrestrial receptors (e.g., insectivorous, benthivorous, piscivorous, and carnivorous birds, and insectivorous, semi-piscivorous, and piscivorous mammals) that prey on lake biota.

A detailed summary of the 1992 information used in this BERA is presented in Table 7-1. Station locations are presented in Figures 7-1 to 7-5. Detailed descriptions of sampling and analytical methods are presented in PTI (1993b,c,d,e). All 1992 data used in this BERA are located in Appendix I.

7.2 1999 and 2000 Field Investigations

Following the submittal of the draft BERA in May 1998, a supplemental field investigation was conducted by Honeywell/Exponent during 1999 and 2000 (Exponent, 2000) to collect additional information needed for the BERA, the HHRA, and the RI, as requested by NYSDEC. A detailed summary of the investigations used in the BERA is presented in Table 7-2. Station locations are presented in Figures 7-6 to 7-10.

A limited amount of water column sampling was conducted in 1999 to evaluate conditions during fall turnover at stations in the centers of both basins of the lake and to evaluate water quality from a human health perspective at nine nearshore stations. In 2000, supplemental sampling of lake sediment, sediment porewater, wetlands sediment, dredge spoils area soil, and biota (benthic organisms, young-of-year [YOY] fish, and adult fish) was performed. All 1999 and 2000 data used in this BERA are included in Appendix I.

7.3 Other Sources of Information

In addition to information collected by Honeywell during the RI, relevant information collected by other parties was also used in the BERA. Major sources of such information include:

- Sediment/soil data collected from five locations (two depths per location, 0 to 15 cm, 15 to 30 cm) by NYSDEC/TAMS in May 2002 in Wetland SYW-6 adjacent to Station S375 sampled in 2000 (Figure 7-10) (TAMS, 2002b).

- Monitoring data on water chemistry, phytoplankton, and zooplankton collected by the Onondaga County Department of Water Environment Protection (OCDWEP) (Stearns & Wheler, 1994; OCDWEP, 2002).

- Data on aquatic macrophytes collected by the US Army Corps of Engineers (USACE) (Madsen et al., 1993, 1996, 1998; Auer et al., 1996a).

- Data on fish communities collected by researchers at State University of New York College of Environmental Science and Forestry (SUNY ESF) (Gandino, 1996; Ringler et al.,1995; Auer et al., 1996a; Tango and Ringler 1996).

- Fish tissue data collected by NYSDEC (1992 to 2000, unpublished).

- Data on amphibians and reptiles collected by researchers at SUNY Cortland (Ducey and Newman, 1995; Ducey, 1997; Ducey et al., 1998, 2000).

- Data on zooplankton conducted by researchers at Cornell University (Hairston et al. 1999; Duffy et al. 2000).

Analytical data used in this BERA are located in Appendix I.

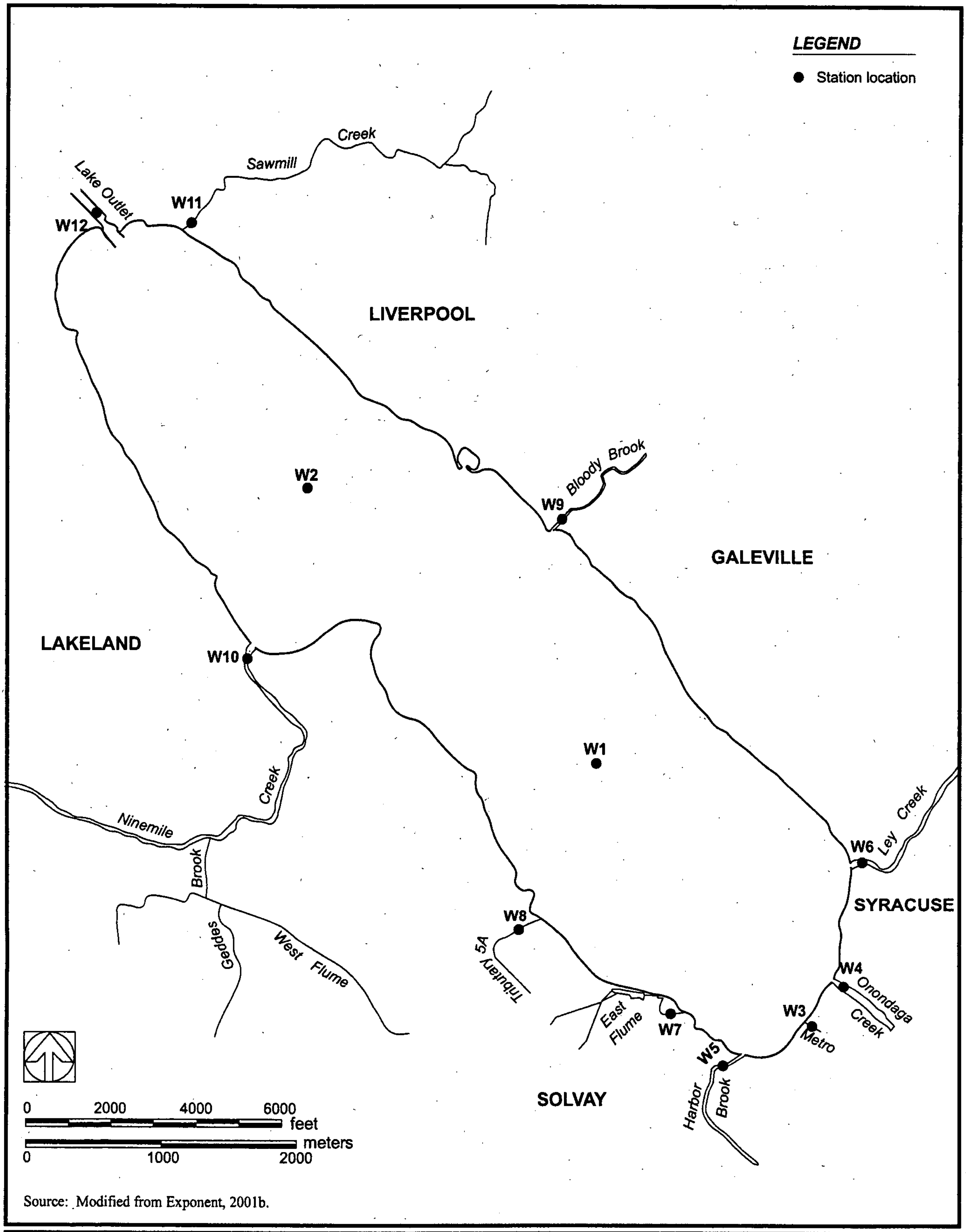
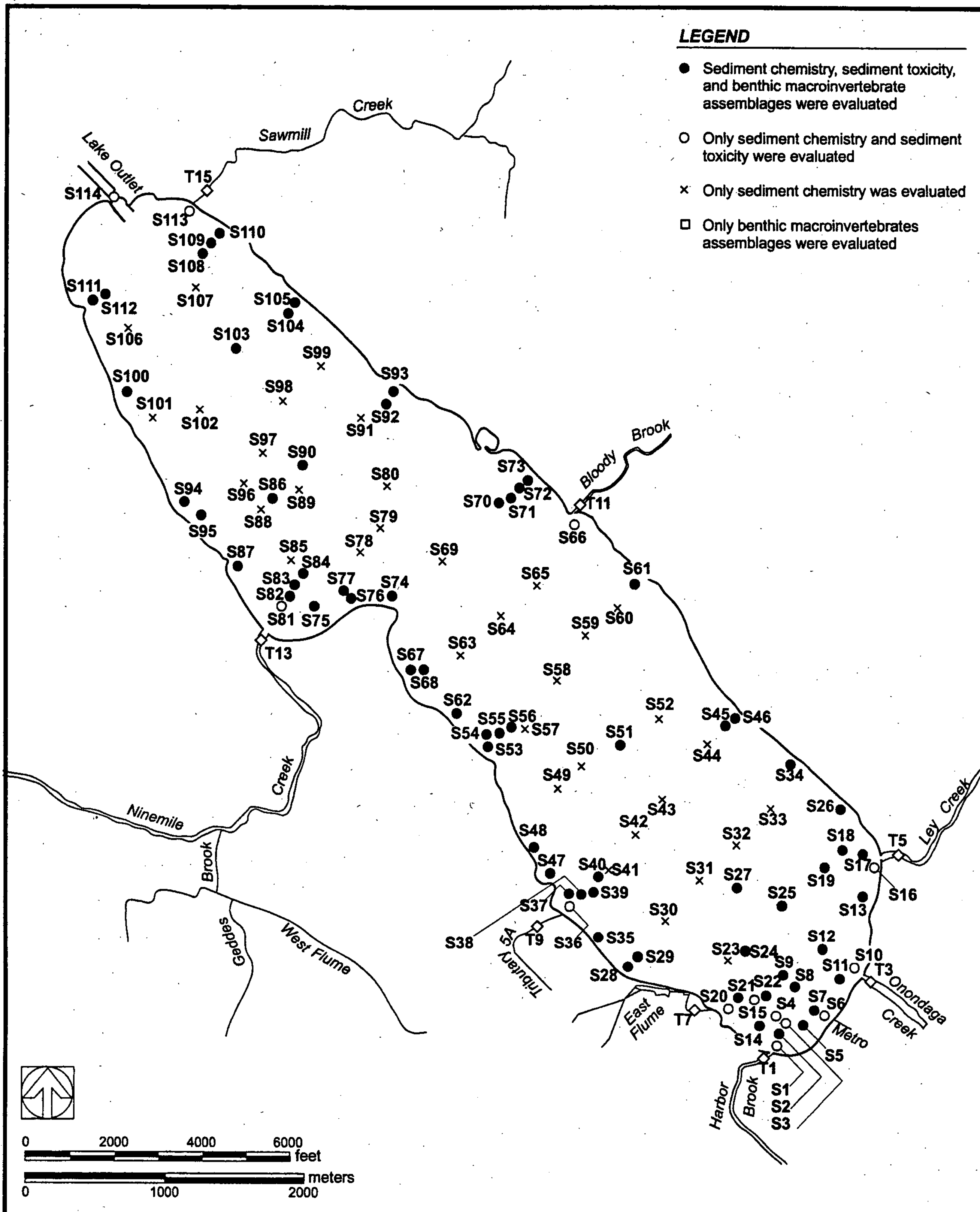
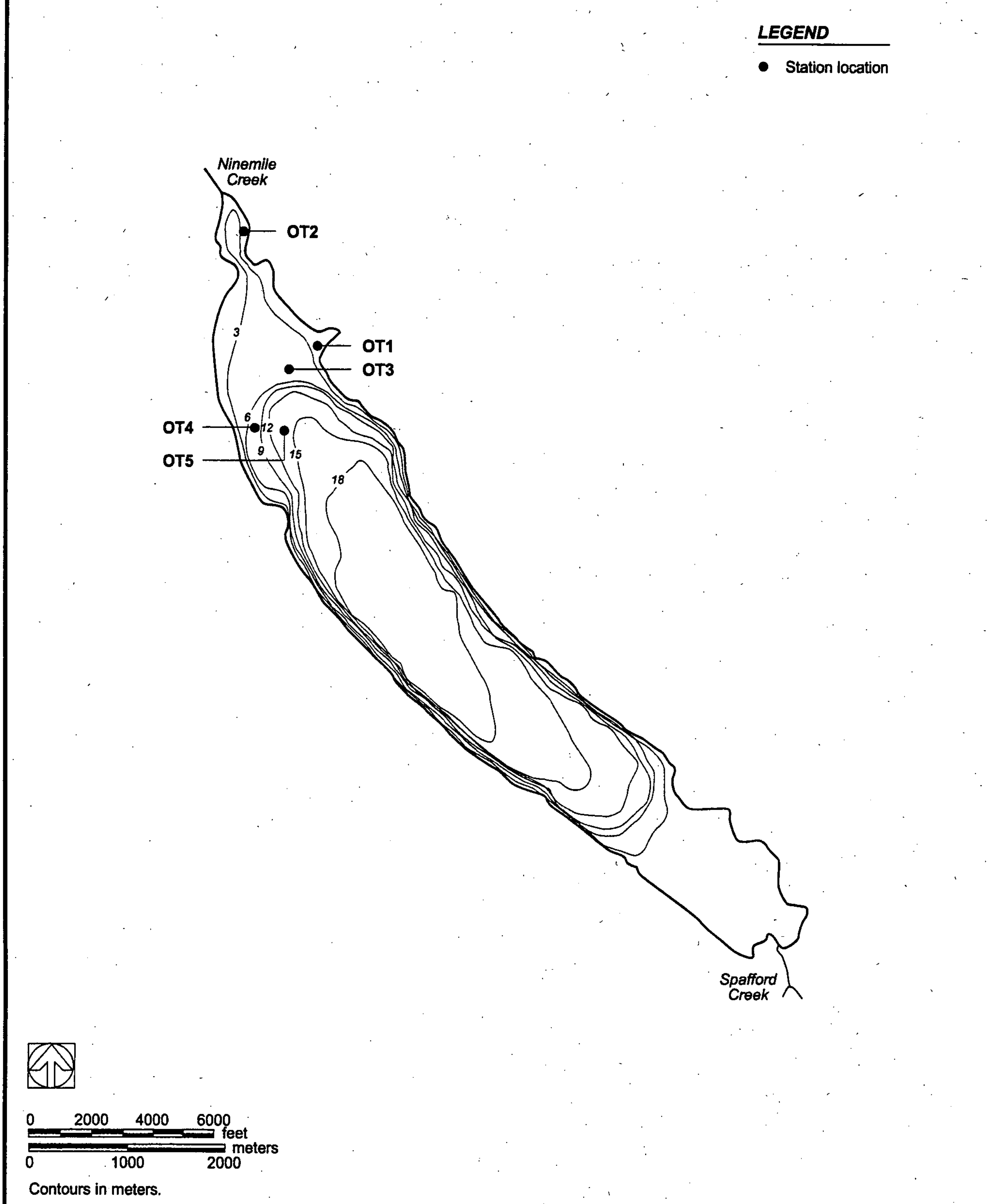


Figure 7-1: Locations of Stations at which Water Samples were Evaluated in Onondaga Lake, its Tributaries, and the Metro Outfall during the 1992 Sampling



Source: Modified from Exponent, 2001b

Figure 7-2. Locations of Stations at which Sediment Chemistry, Sediment Toxicity, and Benthic Macroinvertebrate Assemblages were Evaluated in Onondaga Lake and its Tributaries During the 1992 RI Sampling



Source: Modified from Exponent, 2001b.

Figure 7-3. Locations of Stations at which Sediment Chemistry, Sediment Toxicity, and Benthic Macroinvertebrate Assemblages were Evaluated in Otisco Lake During the 1992 RI Sampling

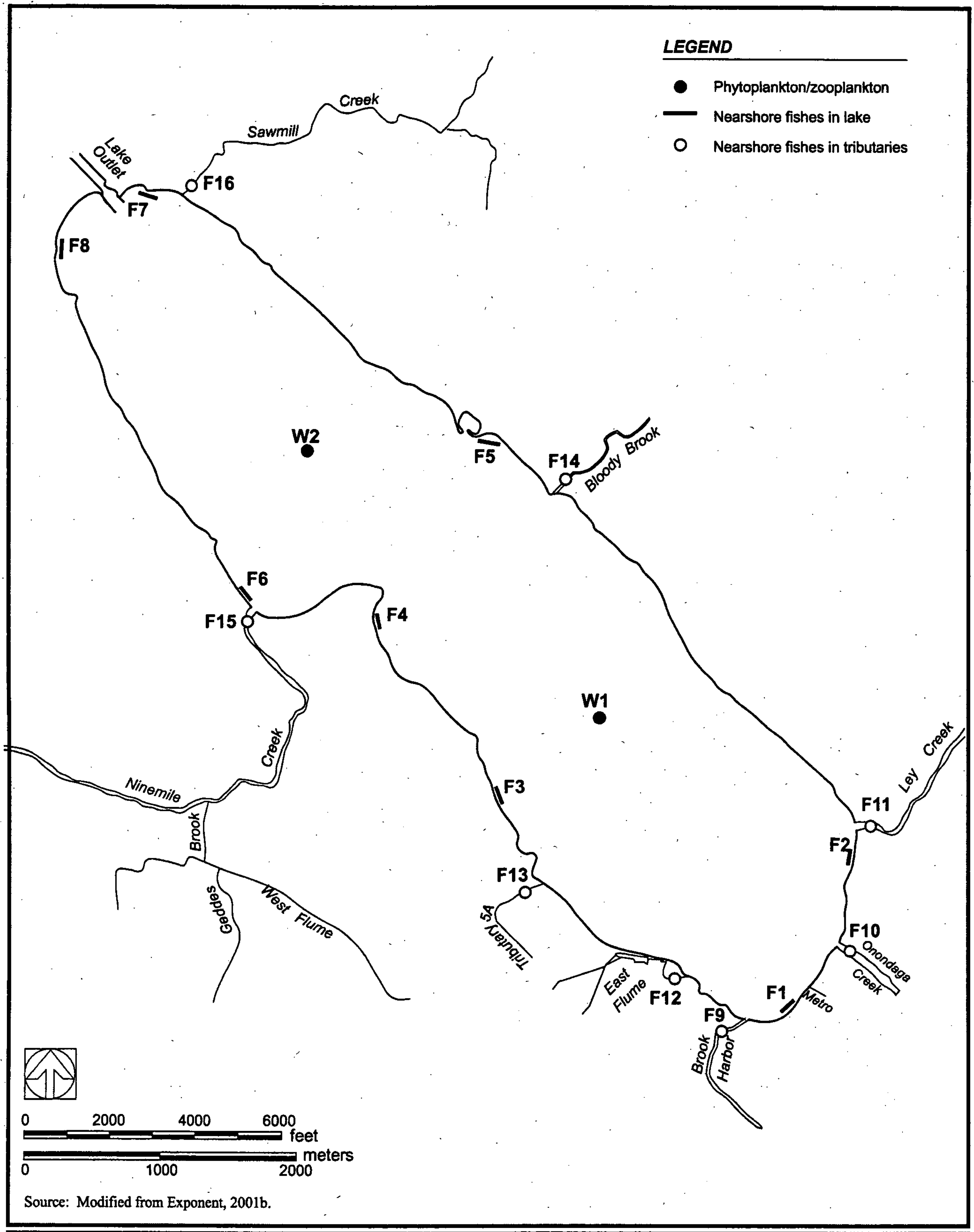


Figure 7-4. Locations of Stations at which Phytoplankton, Zooplankton, and Nearshore Fish Assemblages were Evaluated in Onondaga Lake and its Tributaries During the 1992 RI Sampling

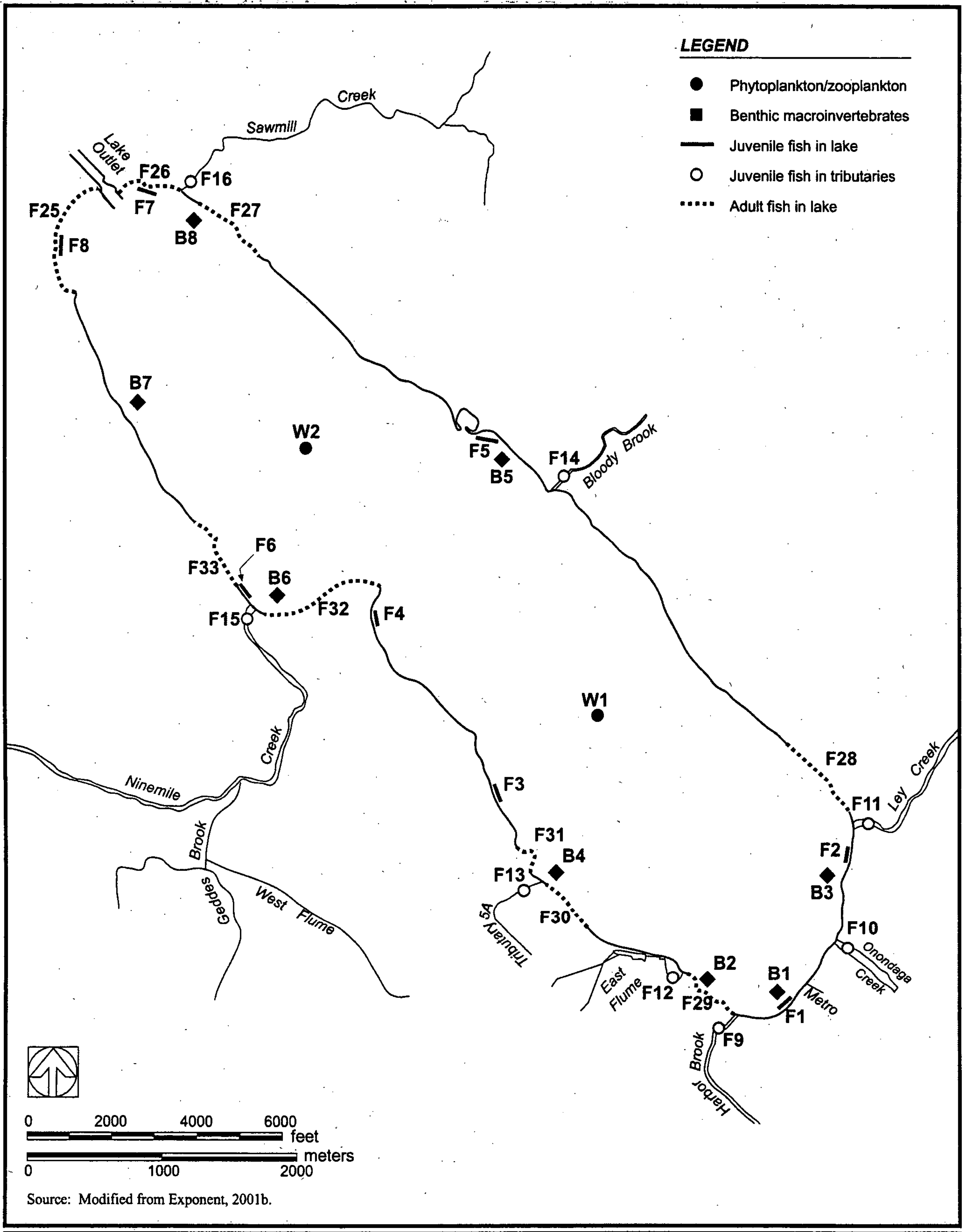


Figure 7-5. Locations of Stations at which Bioaccumulation in Phytoplankton, Zooplankton, Benthic Macroinvertebrates, and Fish were Evaluated in Onondaga Lake and its Tributaries During the 1992 RI Sampling

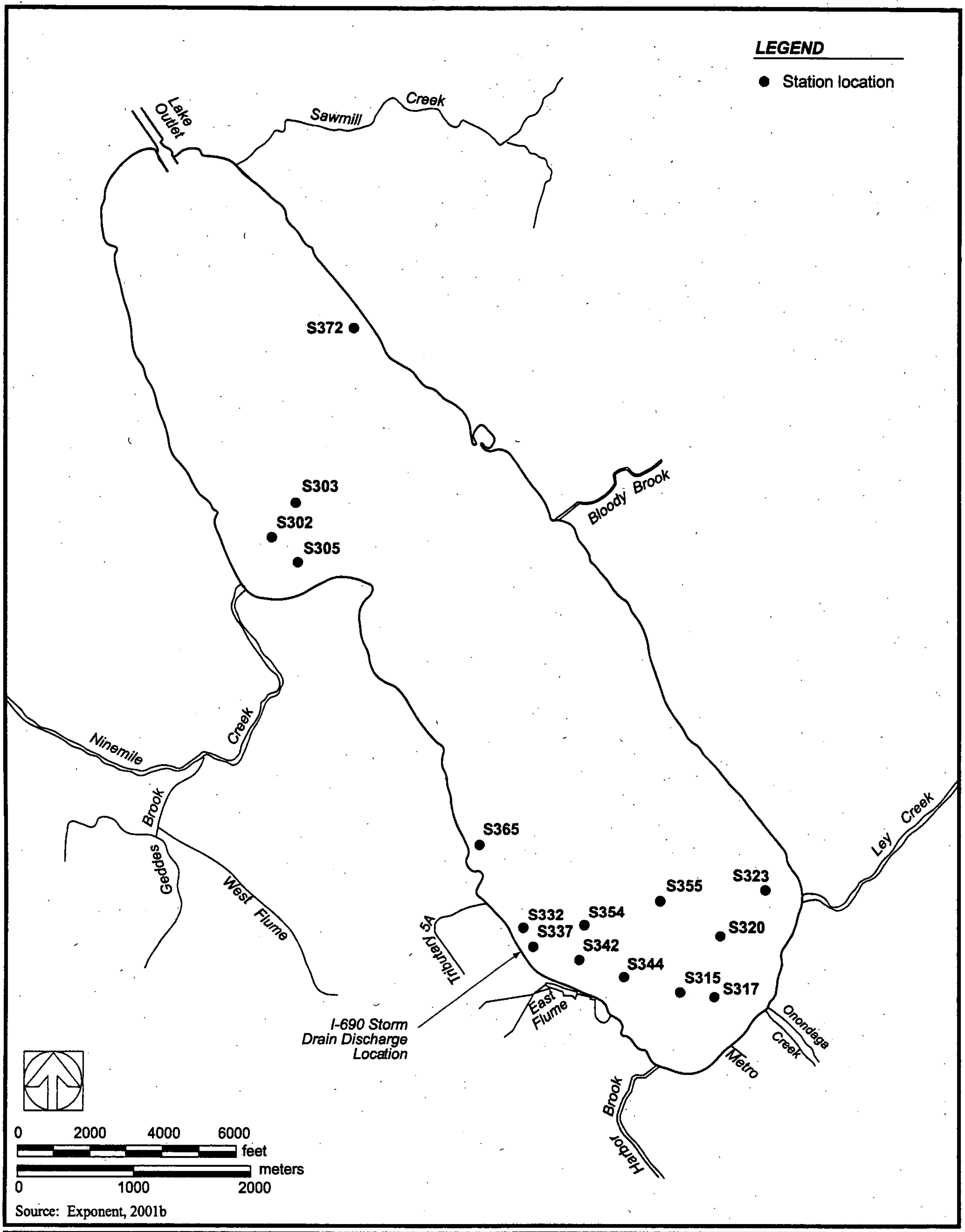
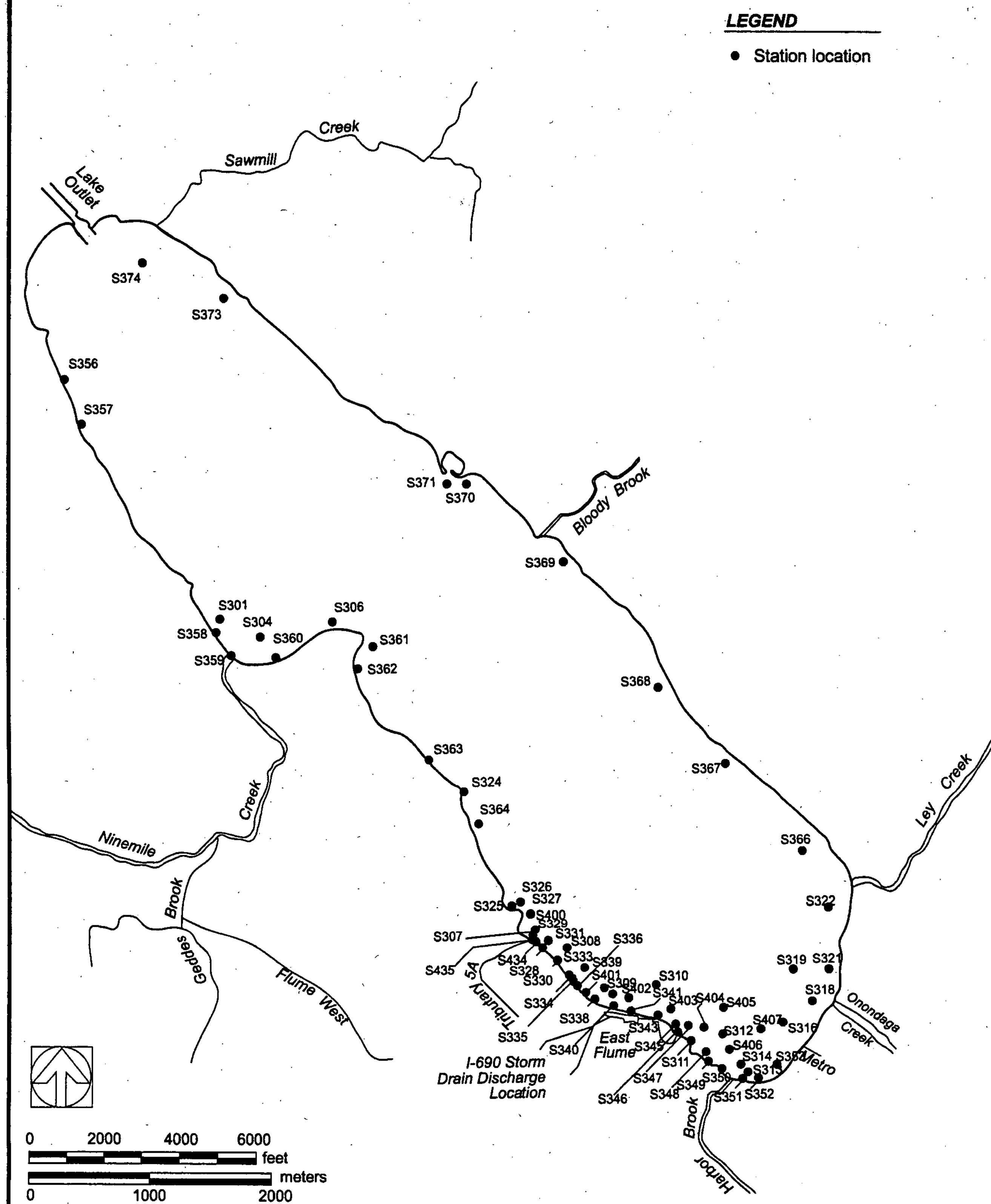


Figure 7-6. Locations of stations at which sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities were evaluated in Onondaga Lake in 2000



Source: Exponent, 2001b.

Figure 7-7. Locations of stations at which only sediment chemistry was evaluated in Onondaga Lake in 2000

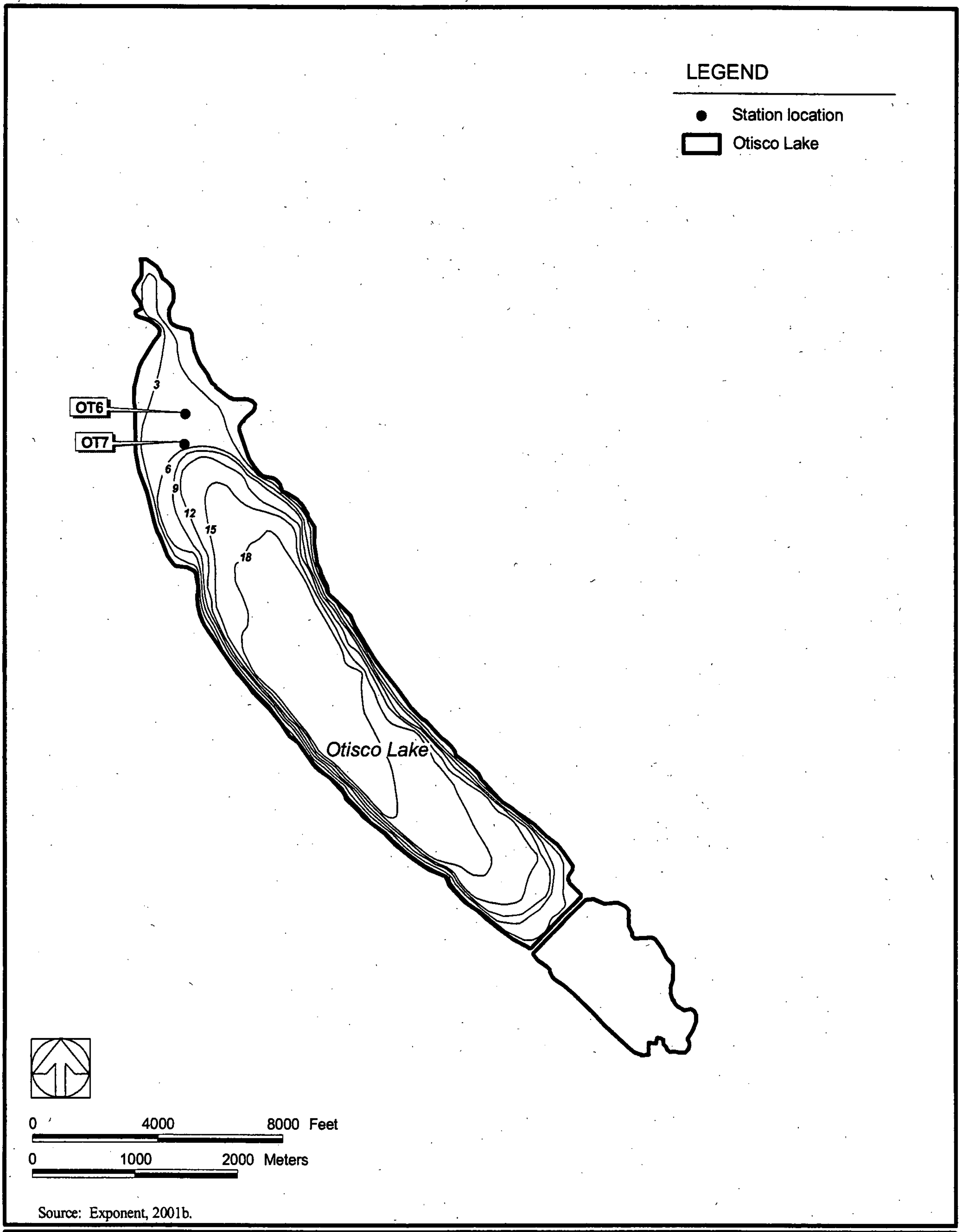


Figure 7-8. Locations of stations at which sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities were evaluated in Otisco Lake in 2000

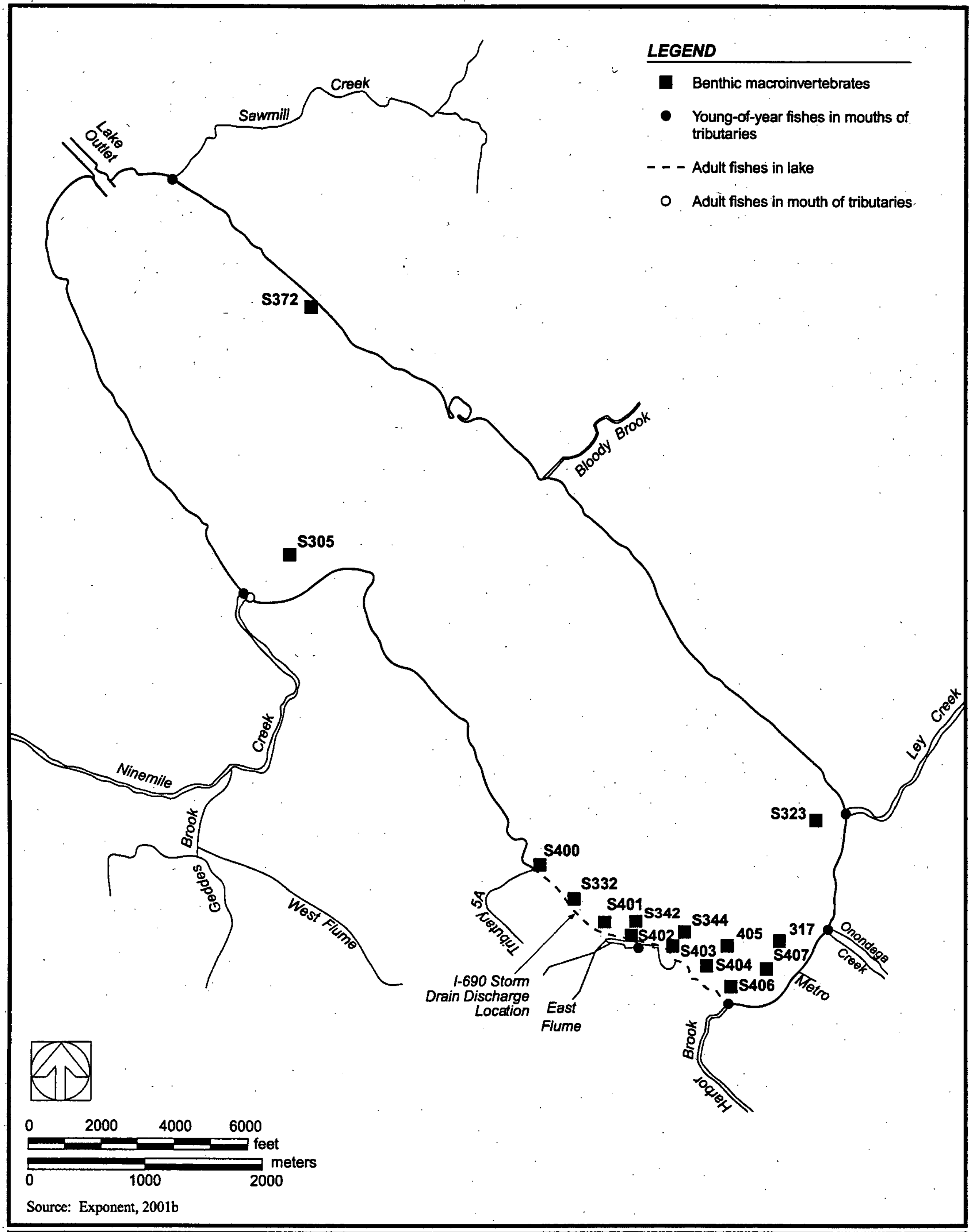


Figure 7-9. Locations of stations at which bioaccumulation in benthic macroinvertebrates and fishes was evaluated in Onondaga Lake and its tributaries in 2000

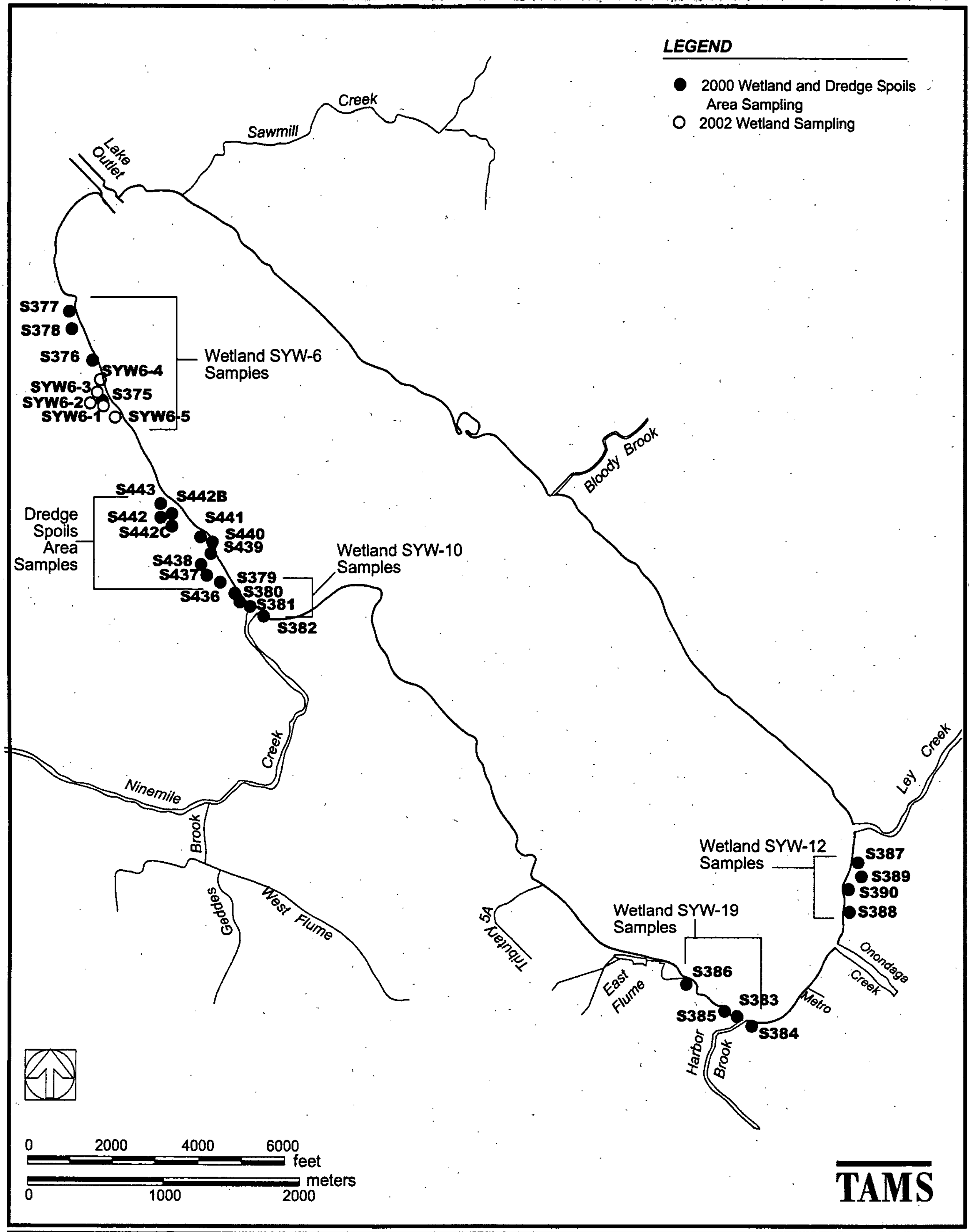


Figure 7-10. Locations of Wetland Sediment and Dredge Spoils Area Soil Stations, 2000 and 2002

Table 7-1. Summary of 1992 Honeywell RI Data Used in the Onondaga Lake BERA

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Mercury and Calcite Mass Balance Investigations					
External Loading and Flushing Study					
Primary tributaries (Onondaga Creek, Harbor Brook, Ley Creek, East Flume, Tributary 5A, Ninemile Creek, and the lake outlet) and Metro outfall After September, site VOCs were dropped from the analytical suite for low-flow samples, except at Harbor Brook, the East Flume, and Tributary 5A	10	1 – low flow 2 – high flow	Twice per month April – Dec.	195	Field measurements pH Temperature Dissolved oxygen Ammonia Chloride Site metals [a] Methylmercury Site VOCs [b] Hexachlorobenzene
Secondary tributaries (Bloody Brook and Sawmill Creek)	2	1	Once per month May (low flow) Dec. (high flow)	4	Field measurements pH Temperature Dissolved oxygen Ammonia Chloride Site metals Methylmercury Site VOCs Hexachlorobenzene

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Water Column Processes Study					
Onondaga Lake water Unfiltered water samples from depths of 0, 3, 6, 9, 12, 15, and 18 m during summer stratification (May – Sept.). Unfiltered water samples from depths of 3, 9, and 15 m during turnover and winter stratification (April, Oct., Nov.). After September, site VOCs were dropped from the analytical suite.	2 (plus duplicates at Station W1 through July)	7 for 5 months 3 for 3 months	Monthly April – Nov.	112	Field measurements pH Temperature Dissolved oxygen Site metals Site VOCs Methylmercury Sulfide Ammonia Chloride
Onondaga Lake water Filtered water samples from depths of 0, 3, 6, 9, 12, 15, and 18 m during summer stratification (May – Sept.). Filtered water samples from depths of 3, 9, and 15 m during turnover and winter stratification (April, Oct., Nov.).	2 (plus duplicates at Station W1 through July)	7 for 5 months 3 for 3 months	Monthly April – Nov.	112	Total mercury Methylmercury
Sediment Processes Study – Nutrients					
Sediment cores to 20 cm (porewater fraction analyzed)	6	4–6	Aug., Nov.	83	Ammonia Hydrogen sulfide
Substance Distribution Investigation					
Lake Water Chemistry Study					
Onondaga Lake Unfiltered water samples from epilimnion and hypolimnion	2	2	Sept.	4	TAL and TCL chemicals

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Sediment Chemistry Study					
Onondaga Lake Full characterization – surface sediments (0 to 2 cm)	19	1	July – Aug.	29	Conventional analytes AVS Calcium carbonate TOC Grain size TAL and TCL chemicals SEM [c] (selected stations)
Partial characterization – surface sediments (0 to 2 cm)	95	1	July – Aug.	95	Conventional analytes AVS Calcium carbonate TOC Grain size Site metals SEM (selected stations) Site VOCs Chlorinated benzenes [d] PAHs (selected stations) PCBs
Otisco Lake Full characterization – surface sediments (0 to 2 cm)	5	1	July – Aug.	5	Conventional analytes AVS Calcium carbonate TOC Grain size TAL and TCL chemicals SEM (selected stations)

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Ecological Effects Investigation					
Sediment Toxicity Study					
Onondaga Lake	79	1	July – Aug.	83	Amphipod test Survival Biomass Chironomid test Survival Biomass (At stations S1 and S17 two additional replicates were collected per location)
Otisco Lake	5	1	July – Aug.	5	Amphipod test Survival Biomass Chironomid test Survival Biomass
Benthic Macroinvertebrate Study					
Onondaga Lake	66	5 replicates	July – Aug.	330	Species abundance
Tributaries (1 pool per tributary)	8	5 replicates	Aug.	40	Species abundance
Otisco Lake	5	5 replicates	July – Aug.	25	Species abundance
Nearshore Fish Study					
Littoral zone	8	5 replicates	June/July, Aug./Sept., and Oct./Nov.	120	Species abundance Total length Abnormalities
Tributaries	8	1	June – Nov.	24	Species abundance Total length Abnormalities

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Macrophyte Distribution Study					
Aerial and visual surveys	Entire littoral zone	NA	July – Aug.	NA	Species distribution
Bioaccumulation Investigation					
Phytoplankton Study					
Composite samples at each station from 0, 3, 6, and 12 m	2	3	May, Aug. and Nov.	18	Species abundance Biomass
Zooplankton Study					
Composite samples of entire assemblage and most abundant large zooplankton taxon (cladocerans)	2	3 (assemblages)	May, Aug. and Nov.	18 (assemblages)	Species abundance Biomass (assemblages)
Benthic Macroinvertebrate Study					
Composite samples of amphipods and chironomids at each station	7 (amphipods) 8 (chironomids)	1	Aug.	15	Biomass Methylmercury
Fish Tissue Study					
Fillets from individual adults from the northern, southern, and western parts of Onondaga Lake					
Gizzard shad	2	10	Aug. – Sept.	20	Total length
Carp	2	10	Aug. – Sept.	20	Biomass
Channel catfish	2	10–11	Aug. – Sept.	21	Age
White perch	2	10	Aug. – Sept.	20	Sex
Bluegill	3	10	Aug. – Sept.	30	Abnormalities
Smallmouth bass	3	10	Aug. – Sept.	30	Methylmercury
Walleye	2	9–11	Aug. – Sept.	20	PCBs Percent lipids

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses
Composite samples of fillets from four individual adults from the southern part of Onondaga Lake					
Channel catfish	1	1 composite	Sept. – Oct.	1	Total length
White perch	1	1 composite	Sept. – Oct.	1	Biomass
Smallmouth bass	1	1 composite	Sept. – Oct.	1	Age
Walleye	1	1 composite	Sept. – Oct.	1	Sex
					Abnormalities [e]
					TAL and TCL chemicals
					Percent lipids
Whole bodies of individual adults from the northern and southern parts of Onondaga Lake					
Gizzard shad	2	5	Aug.	10	Total length
White perch	2	5	Aug.	10	Biomass
Bluegill	2	5	Aug. – Sept.	10	Age
Smallmouth bass	2	5	Aug. – Sept.	10	Abnormalities
					Methylmercury
					Percent lipids
Composite samples of whole bodies of 5 to 12 individual juveniles					
Most abundant species in littoral zone	8	1–2 composites	Aug.	10	Total length
					Biomass
					Abnormalities
					Methylmercury
					PCBs
					Percent lipids
Most abundant species in tributaries	7	1–2 composites	Sept.	7	

Notes:
AVS – acid-volatile sulfides
BTEX – benzene, toluene, ethylbenzene, and xylenes
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RI – remedial investigation
SEM – simultaneously extracted metals
TAL – USEPA's Target Analyte List for inorganic chemicals

Table 7-1. (cont.)

Investigation/Study	No. Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples	Analyses

TCL – USEPA's Target Compound List for organic chemicals
TOC – total organic carbon
VOC – volatile organic compound

[a] Site metals: cadmium, chromium, copper, lead, mercury, nickel, and zinc.
[b] Site VOCs: BTEX compounds and mono-, di-, and trichlorobenzenes.
[c] SEM: cadmium, copper, lead, mercury, nickel, and zinc.
[d] Chlorinated benzenes: tetra-, penta-, and hexachlorobenzenes.
[e] No abnormalities were subsequently found in any of the fishes collected for the RI in 1992 (PTI, 1993c).

Table 7-2. Summary of 1999 and 2000 Honeywell RI and 2002 NYSDEC Data Used in the Onondaga Lake BERA

Investigation/Study	No. of Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples [a]	Analyses
Sediment Investigation					
Surface Sediment (0–15 cm)					
Onondaga Lake	84	1–9	7/13–8/13/00	157	Chemical analysis
Otisco Lake	2	1	8/9 and 8/14/00	2	Chemical analysis
Porewater (0–8 cm)					
Onondaga Lake	7	6	July 2000	42	Mercury, Methylmercury, pH, TOC and Total Solids
Wetland Sediment (0–15 cm)					
Onondaga Lake	21	1	8/11–8/13/00 and 5/9/02 [b]	21	Chemical analysis
Dredged Material					
Onondaga Lake	8	5–6 [c]	8/16–8/17/00	41	Chemical analysis
Water Column Investigation					
Onondaga Lake	12	1–35	9/27/99–12/2/99	73	Chemical analysis
Aquatic Ecological Investigation					
Whole Fish, Fillets, Remainders, and Composites					
Onondaga Lake	9	1-22 [d]	9/19–9/22/00	55	Species abundance Total length Biomass Age Chemical analysis

Table 7-2. (cont.)

Investigation/Study	No. of Stations	No. Samples per Station	Sampling Period(s)	Total No. Samples [a]	Analyses
Benthic Macroinvertebrates (0–15 cm)					
Onondaga Lake	15	5 replicates	8/10–8/13/00	75	Species abundance
Otisco Lake	2	5 replicates	8/9 and 8/14/00	10	Species abundance
Onondaga Lake	15	2–4	7/28–8/2/00	41	Chemical analysis of major taxa [e]
Otisco Lake	2	2 or 3	8/3/00	5	Chemical analysis of major taxa [f]
Sediment Toxicity (0–15 cm)					
Onondaga Lake	15	1	8/10–8/13/00	15	Amphipod test: survival, biomass, reproduction
Otisco Lake	2	1	8/9 and 8/14/00	2	Amphipod test: survival, biomass, reproduction

Notes: [a] The number of samples does not include field quality control samples (i.e., field duplicate and field replicate samples).

[b] The May 2002 wetland sampling was conducted by NYSDEC/TAMS and accounted for five samples in Wetland SYW-6.

[c] The presence of distinct layers observed in dredged material occasionally increased the number of intervals collected in a core.

[d] Whole adult fish were collected near and in the mouth of Ninemile Creek and near the shore of Onondaga Lake from Tributary 5A to Harbor Brook. Composite young-of-the-year fish were collected from the mouths of Ninemile Creek, East Flume, Ley Creek, Harbor Brook, Onondaga Creek, Sawmill Creek, and Bloody Brook.

[e] Benthic macroinvertebrates utilized for chemical analysis were collected from 7/28/00 through 8/2/00 at 15 stations; however, as specified in the work plan (Exponent, 2000d), these were not the same 15 stations that were used for analysis of species abundance and toxicity testing.

[f] Benthic macroinvertebrates utilized for chemical analysis were collected on 8/3/00 at the reference area.

8. ANALYSIS OF ECOLOGICAL EXPOSURES (ERAGS STEP 6)

8.1 Chemical and Stressor Characterization

In this section, the distributions of chemicals of concern and stressors of concern (COCs and SOCs) in Onondaga Lake media are described. Most of the information used in this section to characterize the general distributions of COCs and SOCs in the lake was taken from the 1992 and 1999/2000 field surveys conducted by Honeywell, and additional data (e.g., NYSDEC fish data from 1992 to 2000 and 2002 wetland sediment/soil data) were used. Additional detail on the nature and extent of contamination can be found in Chapter 5 of the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

8.1.1 Distribution of Chemicals and Stressors of Concern in Water

In this section, the distributions of COCs/SOCs in the water of Onondaga Lake and its tributaries are described. Detailed summary tables of COC/SOC concentrations in water are presented in Appendix B, with 1992 data presented in Tables B-1 through B-26 and 1999 data presented in Tables B-27 through B-31. The 1992 data set is more extensive than the supplemental sampling performed in 1999. In 1992, sampling was conducted monthly from April to November at one station in the northern basin and one station in the southern basin of the lake. The 1999 sampling was oriented toward collecting data for the Onondaga Lake Human Health Risk Assessment (HHRA) (TAMS, 2002a) and included sampling selected areas where people could be exposed to lake water. Samples were also collected monthly from September to December 1999 at the southern basin station (W1) and in September and October 1999 at the northern basin station (W2).

In addition, 1997 to 2001 data for eutrophic stressors from the Onondaga County Ambient Monitoring Program (Onondaga County Department of Water Environment Protection [OCDWEP], 2002a, Onondaga County Department of Drainage and Sanitation [OCDDS], 1998]) are presented in this BERA. This program is intended to monitor Onondaga Lake, its tributaries, and the Seneca River in order to evaluate the impacts of alterations and improvements to the Metropolitan Syracuse Sewage Treatment Plant (Metro) and the combined sewer overflows (CSOs) on water quality. The program includes:

- Onondaga Lake monitoring.
- Tributary monitoring.
- Storm event monitoring in Onondaga Lake, Onondaga Creek, Ninemile Creek, Harbor Brook, and Ley Creek.
- Seneca River monitoring.
- Macroinvertebrate sampling in Onondaga Creek, Harbor Brook, and Ley Creek.

These areas are to be monitored from 1999 until 2012. Improvements to Metro and the CSOs are to be implemented in a phased program through this time period under an Amended Consent Judgment (ACJ) entered into on January 20, 1998, with a final completion date for the improvement projects in the year 2012. These improvements are designed to reduce loading of wastewater-related pollutants (i.e., ammonia, phosphorus, solids, floatables, and bacteria) by improvements to the CSO, filtration, and monitoring systems.

Monitoring stations in Onondaga Lake are at the northern and southern deep basins and, for storm event sampling, in nearshore areas of the lake, including the mouth of Ninemile Creek, the mouth of Sawmill Creek, the mouth of Ley Creek, the mouth of Harbor Brook, off of the Metro outfall, in the northeast corner of the lake between Bloody Brook and Sawmill Creek, and the northwest corner of the lake near the lake outlet.

Data from the ambient water quality monitoring program from 1997 through the first quarter of 2001 are included in select figures in this chapter.

The lake monitoring includes:

- Biweekly profiles of field parameters (pH, temperature, dissolved oxygen [DO], specific conductance, oxidation-reduction potential, salinity, and conductivity) at 0.5 m depth intervals at the southern deep station for the entire monitoring period (April through November).

- Biweekly profiles of ammonia, total Kjeldahl nitrogen (TKN), nitrate (N), and organic nitrogen at 3 m intervals at the southern deep station for the entire period and in the winter when possible.

- Biweekly samples of total phosphorus (P) at 1 m depths at the southern deep station from June through September.

- Biweekly analysis of solids and organic and inorganic carbon at 6 m intervals in the southern deep station.

- Biweekly composite samples of total dissolved solids in the epilimnion and the hypolimnion at the southern deep station.

- Biweekly Secchi disk transparency measurements at the southern basin for the entire period and at nearshore stations from June through September.

- Also included are DO profiles at 0.5 m depth intervals at the northern and southern deep stations during fall mixing (to include other water quality parameters during mixing), as well as DO measurements taken at tributary mouths once during mixing, and diurnal sampling for DO, pH, and temperature in Onondaga Lake upper waters at 3-hour intervals for a 24-hour period during warm weather and algal bloom conditions.

In 1992, surface water samples were collected from eleven tributaries in the Onondaga Lake area in order to aid in determining the nature and extent of contamination and contaminant loadings to the lake. Most metal COCs (other than mercury) were detected occasionally in tributaries and point sources. Cadmium was detected infrequently, while copper, lead, and zinc were frequently detected. Tributary 5A, Geddes Brook, and the East Flume are the three tributaries in which the majority of the metal COCs were more frequently detected.

Organic COCs were occasionally detected in Tributary 5A, the East Flume, Harbor Brook and Geddes Brook. With the exception of a single detection of toluene in Metro effluent, organic COCs were not detected in any other tributaries in 1992. The tributaries and Metro effluent were also sampled for several other SOCs (chloride, ammonia, and depleted DO). The 1992 surface water data were stratified into three flow regimes: base flow, high flow, and intermediate flow. Base-flow conditions were determined by examination of daily flow records for each tributary, and were generally set to low flows in the late summer and early fall. High-flow conditions were defined as the highest 10 percent of the average daily flows, and intermediate flows were defined as flows between base-flow and high-flow stages. Summary statistics for the 1992 tributary data for base-flow, intermediate-flow, and high-flow conditions, as well as available 1992 and 1999 lake data, are presented in Tables 8-1, 8-2, and 8-3.

8.1.1.1 Mercury

Lake Water

The concentrations of total mercury and methylmercury found in unfiltered water samples from the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 and September to December 1999 are presented in Figures 8-1 and 8-2, respectively. A high-flow event in April 1992 accounted for the highest concentration of total mercury (29 ng/L), which was detected in the southern basin hypolimnion (15 m depth).Otherwise, total mercury concentrations increased from spring to fall in both the epilimnion and hypolimnion. The increase in the hypolimnion was greater than the increase in the epilimnion. Methylmercury concentrations in the hypolimnion increased substantially from spring to fall of 1992, rising from a lake average of 1.0 ng/L in May to a maximum of 12 ng/L in early October. Methylmercury concentrations in the epilimnion increased at a higher rate than total mercury concentrations over the course of the year, rising from 0.4 ng/L in May to 1.5 ng/L in November, as compared to the rise in total mercury from 3.7 ng/L to 7.4 ng/L.

The mixing of lake water during fall turnover resulted in mercury and methylmercury concentrations in the hypolimnion declining and concentrations in the epilimnion increasing, so that contaminants were fairly evenly distributed at various lake depths. The maximum and mean concentrations of mercury detected in lake water in 1992 of 29 and 9.0 ng/L were above the NYSDEC wildlife water quality value of 2.6 ng/L dissolved concentration (Chapter 4, Table 4-4; Appendix B, Table B-4).

In 1999, mercury in lake water was also analyzed from September to December in the southern basin and in September and October in the northern basin (Appendix B, Table B-27). Sampling was also performed at nearshore locations around the lake (see Chapter 2, Figure 2-17 of the RI). The maximum and mean concentrations of mercury detected in lake water in 1999 of 103 and 11 ng/L were above the NYSDEC wildlife water quality value of 2.6 ng/L (Appendix B, Table B-27). The maximum value of 103 ng/L was detected at Station W55 near Harbor Brook. A comparison of the 1992 and 1999 data showed no consistent pattern of increases or decreases of total mercury or methylmercury seasonally or by depth, indicating that mercury concentrations have remained elevated in Onondaga Lake over the last decade.

Tributary Water and Metro Discharge

Mean concentrations of total mercury and methylmercury in tributary water and Metro effluent in 1992 are presented for base-flow, intermediate-flow, and high-flow conditions in Figures 8-3 and 8-4.

Under base-flow conditions, mean concentrations of total mercury were highest in Geddes Brook (100 ng/L), the East Flume (54 ng/L), Metro effluent (24 ng/L), and lower Ninemile Creek (13 ng/L), and were less than 10 ng/L elsewhere. Mean total mercury concentration in Ninemile Creek downstream of the Geddes Brook confluence (13 ng/L) was significantly higher than the mean concentration in upper Ninemile Creek (3.8 ng/L) under base-flow conditions. Mean concentrations of methylmercury were highest in the Metro effluent (2.1 ng/L), the East Flume (0.8 ng/L), Harbor Brook (0.4 ng/L), and the lake outlet (0.4 ng/L), and were less than 0.3 ng/L elsewhere.

Under intermediate-flow conditions, mean concentrations of total mercury were highest in Geddes Brook (154 ng/L), the East Flume (96 ng/L), Tributary 5A (36 ng/L), and lower Ninemile Creek (31 ng/L), and concentrations were less than 25 ng/L elsewhere. Mean concentrations of methylmercury under intermediate flow conditions were highest in the East Flume (1.6 ng/L), the lake outlet (1.2 ng/L), Metro effluent (1.0 ng/L), Harbor Brook (0.61 ng/L), Onondaga Creek (0.54 ng/L), Tributary 5A (0.53 ng/L), and Geddes Brook (0.50 ng/L), and less than 0.5 ng/L elsewhere.

Under high-flow conditions, mean concentrations of total mercury were highest in the East Flume (155 ng/L), Geddes Brook (145 ng/L), Tributary 5A (94 ng/L), Harbor Brook (41 ng/L), Metro effluent (39 ng/L), and lower Ninemile Creek (27 ng/L), and were less than 20 ng/L elsewhere. A single high-flow total mercury concentration of 47 ng/L was reported for Bloody Brook. Mean concentrations of methylmercury were highest in Harbor Brook (2.4 ng/L), Tributary 5A (1.5 ng/L), the East Flume (1.3 ng/L), Metro effluent (1.2 ng/L), Geddes Book (1.0 ng/L), Ley Creek (0.85 ng/L), and the lake outlet (0.71 ng/L). Concentrations were less than 0.5 ng/L elsewhere.

8.1.1.2 Other Metals

Lake Water

The concentrations of Target Analyte List (TAL) inorganics were measured in unfiltered water samples from the southern and northern basins of Onondaga Lake in 1992. Analytes that were also analyzed in tributary water are presented in Table 8-1, with the exception of calcium and sodium. A complete list of inorganics and their detection frequency in lake water is provided in Table D-1 in Appendix D. Chromium, lead, and nickel were also sampled at a limited number of locations in 1999. The following distributions were found for each metal in 1992:

- **Barium** – Detected in all samples (n=4), with lakewide maximum and mean concentrations of 77 and 73 µg/L, respectively. All values exceeded the USEPA Tier II water quality criterion of 3.9 µg/L.

- **Cadmium** – Detected in about 4 percent of the samples collected in the lake, with maximum and mean concentrations of 2.9 and 1.0 µg/L, respectively, in the southern basin, and 3.1 and 1.1 µg/L, respectively, in the northern basin. Maximum concentrations in both basins exceeded the USEPA ambient water quality criterion (AWQC) final chronic value (FCV) of 1.3 µg/L and the USEPA and NYSDEC chronic aquatic values of 2.8 and 2.6 µg/L, respectively.

- **Chromium** – Detected in about 15 percent of samples, with maximum and mean concentrations of 5.3 and 1.7 µg/L, respectively, in the southern basin, and 4.1 and 1.5 µg/L, respectively, in the northern basin. All values were below the USEPA and NYSDEC chronic aquatic value of 94.8 µg/L.

- **Copper** – Detected in approximately 32 percent of samples, with maximum and mean concentrations of 51 and 2.5 µg/L, respectively, in the southern basin, and 3.0 and 1.3 µg/L, respectively, in the northern basin. The maximum concentration detected in the southern basin exceeded all USEPA and NYSDEC water quality criteria ranging from 12 to 18 µg/L.

- **Lead** – Detected in about 31 percent of samples, with maximum and mean concentrations of 18 and 1.1 µg/L, respectively, in the southern basin, and 7.7 and 0.9 µg/L, respectively, in the northern basin. Maximum concentrations in the southern and northern basins exceeded USEPA AWQC and NYSDEC chronic water quality values, ranging from 3.5 to5.2 µg/L.

- **Manganese** – Detected in about 98 percent of samples, with lakewide maximum and mean concentrations of 880 and 189 µg/L, respectively. All values exceeded the USEPA Tier II water quality criteria of 80 µg/L.

- **Nickel** – Detected in approximately 12 percent of samples, with maximum and mean concentrations of 15 and 3.7 μg/L, respectively, in the southern basin, and 5.3 and 3.1 μg/L, respectively, in the northern basin. All values were below the USEPA and NYSDEC chronic aquatic value of 67 μg/L.

- **Zinc** – Detected in about 96 percent of samples, with maximum and mean concentrations of 143 and 19 μg/L, respectively, in the southern basin, and 80 and 16 μg/L, respectively, in the northern basin. The maximum concentration in the southern basin was above the NYSDEC chronic water quality value of 107 μg/L and the USEPA AWQC-FCV water quality value of 135 μg/L.

- **Cyanide** – Detected in one of four samples, with a concentration of 171 μg/L in one 6 m sample from the northern basin. This concentration was above the USEPA and NYSDEC chronic and acute water quality values of 5.2 μg/L and 22 μg/L, respectively.

Detection frequencies and contaminant concentrations were generally higher in the southern basin than the northern basin. Chromium and nickel were also detected in the 1999 sampling (Appendix B, Table B-28).

Tributary Water and Metro Discharge

Concentrations of metals other than mercury in tributary water and Metro effluent in 1992 are presented for the three flow regimes in Table 8-1. Mean concentrations for each tributary and Metro are shown in Figures 8-5, 8-6, and 8-7. Under base-flow conditions, the following metal COCs were detected:

- **Cadmium** – Cadmium was undetected in all tributaries.

- **Chromium** – Chromium was detected in five tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (28 μg/L), followed by the lake outlet (18 μg/L), and East Flume (11 μg/L).

- **Copper** – Maximum concentrations in seven tributaries were highest in the East Flume (15 μg/L), Metro effluent (12 μg/L), Tributary 5A (10 μg/L), and Onondaga Creek (6.4 μg/L).

- **Lead** – Lead concentrations were detected in six tributaries, and the maximum concentrations were highest in the East Flume (11 μg/L) and Ley Creek (7.4 μg/L).

- **Nickel** – Nickel was detected in five tributaries and the lake outlet. Maximum concentrations were highest in Sawmill Creek (17 μg/L), the lake outlet (10 μg/L), and Ley Creek (9.5 μg/L).

- **Zinc** – Zinc was detected in eight tributaries, with maximum concentrations in the East Flume (196 μg/L), Tributary 5A (59 μg/L), Onondaga Creek (51 μg/L), and Metro Outfall (42 μg/L).

Under intermediate-flow conditions, the following distributions were found for each metal:

- **Cadmium** – Cadmium was only detected once, in Tributary 5A (2.4 μg/L).

- **Chromium** – Chromium was detected in six tributaries. Maximum concentrations were highest in Tributary 5A (119 μg/L) and lower Ninemile Creek (12 μg/L).

- **Copper** – Copper was detected in seven tributaries. Maximum concentrations were highest in Tributary 5A (30 μg/L) and the East Flume (18 μg/L).

- **Lead** – Lead was detected in seven tributaries and the lake outlet. Maximum concentrations were highest in the East Flume (29 μg/L) and lower Ninemile Creek (22 μg/L).

- **Nickel** – Nickel was detected in four tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (372 μg/L), lower Ninemile Creek (93 μg/L), and the lake outlet (115 μg/L).

- **Zinc** – Zinc was detected in eight tributaries and the lake outlet. Maximum concentrations were highest in the East Flume (179 μg/L), Tributary 5A (117 μg/L), Ninemile Creek (88 μg/L), and Onondaga Creek (85 μg/L).

Under high-flow conditions, the following distributions were found for each metal:

- **Cadmium** – Cadmium was detected in Bloody Brook (17 μg/L), Tributary 5A (3.2 μg/L), and lower Ninemile Creek (2.1 μg/L).

- **Chromium** – Chromium was detected in six tributaries. Maximum concentrations were highest in Tributary 5A (560 μg/L) and Ley Creek (19 μg/L).

- **Copper** – Copper was detected in nine tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (125 μg/L), Ley Creek (58 μg/L), and Harbor Brook (48 μg/L).

- **Lead** – Lead was detected in all tributaries and the lake outlet. Maximum concentrations were highest in Ley Creek (95 μg/L), Tributary 5A (55 μg/L), Harbor Brook (63 μg/L), and Bloody Brook (44 μg/L).

- **Nickel** – Nickel was detected in four tributaries and the lake outlet. Maximum concentrations were highest in Bloody Brook (17 μg/L), Harbor Brook (17 μg/L), and the Metro outfall (15 μg/L).

- **Zinc** – Zinc was detected in all tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (259 μg/L), Bloody Brook (201 μg/L), Harbor Brook (188 μg/L), and Onondaga Creek (182 μg/L).

8.1.1.3 Benzene, Toluene, Ethylbenzene, and Xylenes

Lake Water

No benzene, toluene, ethylbenzene, and xylene (BTEX) compounds were detected in the water of Onondaga Lake in 1992 (Table 8-2).

In 1999, benzene and xylenes were detected in two nearshore areas, and toluene was detected at one nearshore area. The Willis Lakeshore area (Station W50) had the highest benzene concentration at 6.3 μg/L and toluene at 0.16 μg/L. This lake water sample was collected near the groundwater plume originating at the Honeywell Willis Avenue site. The second set of detections were in the lake near Harbor Brook (Station W55), with benzene detected at 0.11 μg/L and total xylenes at 0.33 μg/L. The maximum concentration of xylenes (0.5 μg/L) in the lake was detected near Wastebeds 1 through 8 at Station W53. Ethylbenzene was not detected in the 1999 water samples.

Tributary Water and Metro Discharge

Under base-flow conditions, the following detections of BTEX compounds were found in 1992:

- **Benzene** – Benzene was detected in Harbor Brook (1 to 1.7 μg/L), the East Flume (1.5 μg/L), and Tributary 5A (6.9 to 34 μg/L).

- **Toluene** – Toluene was detected in the Metro effluent (3.1 μg/L), Harbor Brook (1 to 2.6 μg/L), the East Flume (2.5 μg/L), Geddes Brook (5 μg/L), and Tributary 5A (1.1 to 4.2 μg/L).

- **Xylenes** – Xylenes were detected in Harbor Brook (2 to 3.6 μg/L), the East Flume (1.4 μg/L), and Tributary 5A (1.1 to 2.2 μg/L).

There were no detections of ethylbenzene under base-flow conditions in 1992.

There were no detections of benzene, toluene, and xylenes under intermediate-flow conditions in 1992, but ethylbenzene was detected once in Ley Creek (1.7 μg/L).

Under high-flow conditions, the following detections of BTEX compounds were found in 1992:

- **Benzene** – Benzene was only detected in Tributary 5A (60 µg/L). The Semet Residue Ponds and/or the Willis Avenue sites are likely sources of this benzene.

- **Toluene** – Toluene was detected in Tributary 5A (5.1 µg/L) and Harbor Brook (2.1 µg/L).

- **Xylenes** – Xylenes were detected in the East Flume (1.6 to 1.8 µg/L), Tributary 5A (1.1 to 2.4 µg/L), and Harbor Brook (1.7 µg/L).

There were no detections of ethylbenzene under high-flow conditions in 1992.

8.1.1.4 Chlorinated Benzenes

Lake Water

Dichlorobenzenes and trichlorobenzenes were detected in only 1 of 98 water samples collected from Onondaga Lake in 1992 (Table 8-3). That sample was collected from the southern basin. Monochlorobenzene was not detected in any of the 98 samples collected from the lake in 1992.

In 1999, chlorinated benzenes were detected at various locations in nearshore habitat. 1,4-Dichlorobenzene was detected in all samples except the one near the boat ramp in Liverpool. 1,2-Dichlorobenzene was detected in three of 11 samples and monochlorobenzene was detected in only two of 11 samples. For all three compounds, the highest concentrations were detected at the Willis Lakeshore Area (Station W50) at 3.4 µg/L (1,4-dichlorobenzene), 3.2 µg/L (1,2-dichlorobenzene), and 12 µg/L (monochlorobenzene).

Tributary Water and Metro Discharge

Under base-flow conditions, monochlorobenzene (7.6 µg/L), 1,2-dichlorobenzene (2 to10 µg/L), and 1,4-dichlorobenzene (4.6 to13 µg/L) were detected only in the East Flume. Under intermediate flow conditions, 1,4-dichlorobenzene (2.0 to 2.3 µg/L) and 1,2,4-trichlorobenzene (0.9 to1.1 µg/L) were detected only in the East Flume. Under high-flow conditions, monochlorobenzene was not detected in any of the tributaries. Dichlorobenzenes were detected in Harbor Brook (1 µg/L of 1,2-dichlorobenzene and 5.3 to 7.2 µg/L of 1,4-dichlorobenzene) and the East Flume (1 to 5.2 µg/L of 1,2-dichlorobenzene and 7.8 to 20 µg/L of 1,4-dichlorobenzene). Trichlorobenzenes were detected only in the East Flume (1.9 to 2.7 µg/L of 1,2,4-trichlorobenzene).

8.1.1.5 Bis(2-ethylhexyl)Phthalate

The only detections of bis(2-ethylhexyl)phthalate were found in the 1999 lake water samples. The detections were at sample Stations W1 and W2. The four samples analyzed had detections with a maximum concentration of 10 μg/L at Station W2 at a depth of 6 m, with the other three detections of 2 μg/L at Stations W1 and W2 at depths of either 6 or 12 m.

8.1.1.6 Chloride

Lake Water

Chloride concentrations in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-8. During stratification, chloride levels increased in the hypolimnion and decreased in the epilimnion. In the epilimnion, chloride concentrations ranged from 414 to 489 mg/L. The minimum values of chloride concentrations in the epilimnion were found between late June and mid-September, during the period of thermal stratification, with fairly constant concentrations during the remainder of the year.

Chloride concentrations in the hypolimnion were generally greater than the values found in the epilimnion, with a range of 469 to 525 mg/L. In the hypolimnion, the minimum chloride concentration was found in late May. Concentrations then increased continuously during the remainder of stratification. The maximum value was detected in early October. The mean chloride concentration for all samples collected in Onondaga Lake (i.e., epilimnion and hypolimnion) was 485 mg/L.

Tributary Water and Metro Discharge

Concentrations of chloride under base-flow conditions ranged from 46.4 mg/L to 1,411 mg/L. Mean chloride concentrations, shown in Figure 8-9, were highest in lower Ninemile Creek (1,250 mg/L), Onondaga Creek (890 mg/L), and Geddes Brook (730 mg/L), and were less than 500 mg/L in the remaining tributaries. Under intermediate flow conditions, chloride concentrations in the tributaries ranged from 35 to 1,042 mg/L and mean concentrations were greatest in Geddes Brook (682 mg/L) and lower Ninemile Creek (664 mg/L). Under high-flow conditions, chloride concentrations ranged from 36 to 844 mg/L, with the highest mean values occurring in lower Ninemile Creek (670 mg/L) and the East Flume (613 mg/L).

Chloride concentrations and loading (calculated for the USGS Lakeland Station) in Ninemile Creek have been monitored by Honeywell on a quarterly basis since 1989.

8.1.1.7 Salinity

Lake Water

The major components of salinity in the lake are calcium, chloride, and sodium, and minor contributors to lake salinity are magnesium, potassium, bicarbonate (HCO_3-), and sulfate (SO_4) (Effler et al., 1996). Between 1968 and 1990, the volume-weighted salinity was between 2.5 and 3.5 parts per thousand (ppt) (Effler et al., 1996). Relative contributions of calcium, sodium, and chloride decreased markedly with the closure of the soda ash/chlor-alkali facility in 1986, resulting in a drop in the typical salinity of 3.3 ppt in 1981 to 1.1 ppt today (Effler et al., 1996; Onondaga Lake Partnership [OLP], 2002). This value is still an order-of-magnitude greater than the average world river salinity (0.11 ppt) and is several times higher than salinity levels in Otisco Lake (0.25 ppt), whose drainage basin is also within the Limestone Belt of central New York State (Figure 3-8).The salinity level is still artificially high because Solvay waste constituents continue to be released to Onondaga Lake and some of its tributaries from the Solvay Wastebeds located along the lakeshore and Ninemile Creek.

High salinity has altered the biological diversity of Onondaga Lake. Species diversity was reported to be heavily reduced at salinity levels found in Onondaga Lake through 1985 (Remane and Scheiper, 1971). High levels of salinity influenced the phytoplankton, zooplankton, and macrophytes (see Chapter 9, Section 9.1). The reestablishment of some salinity-intolerant zooplankton populations in Onondaga Lake has occurred since the closure of the soda ash/chlor-alkali facility (e.g., Hairston et al., 1999).

The high salinity of Onondaga Lake also affects thermodynamic properties of the lake and the adjoining portions of the Seneca River. Density stratification in the lake has been altered by ion-contaminated inflows to the lake, so that the temperature of maximum density shifted about 0.8°C (1.4°F) lower during winter than previously; the depression has been about 0.3°C (0.5°F) lower since closure (Effler et al., 1996).

Tributary Water and Metro Discharge

Under base-flow conditions, calcium concentrations ranged from 79,600 to 580,000 μg/L, with mean concentrations greatest in lower Ninemile Creek (518,300 μg/L) followed by Geddes Brook (404,200 μg/L). Under intermediate-flow conditions, calcium concentrations ranged from 71,400 to 348,000 μg/L, with lower Ninemile Creek and Geddes Brook averaging 301,600 and 304,000 μg/L, respectively. Calcium concentrations decreased under high-flow conditions in the tributaries, ranging from 54,200 to 383,000 μg/L. Mean high-flow concentrations at lower Ninemile Creek (253,700 μg/L) and Geddes Brook (230,800 μg/L) are lower than the corresponding values under base-flow and intermediate-flow conditions.

8.1.1.8 Nitrogen and Phosphorus

Lake Water

Concentrations of total ammonia, free ammonia, and nitrate (NO_3) that are maintained in the epilimnion throughout productive months and the concentrations of nitrite (NO_2) that develop in the epilimnion are unusually high in Onondaga Lake, compared to concentrations reported for other stratifying lakes in the literature (Effler et al., 1996). The summed concentration of total ammonia and nitrate have continuously exceeded levels associated with limitation of phytoplankton growth, and concentrations of free ammonia and nitrite in the epilimnion routinely exceed the NYSDEC standard of 0.1 mg/L nitrite for warm-water fish to protect non-salmonid (as well as salmonid) fish.

The concentrations of nitrogen species in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-10. These figures show increased concentrations of nitrate and nitrite in the epilimnion during the summer and early fall and decreases in these compounds in the hypolimnion during this period. The lowest concentrations of ammonia were found in the epilimnion from July to August.

More recent sampling of stressors in lake water has been conducted by Onondaga County. Figure 8-11 shows ammonia concentrations in Onondaga Lake measured from 1997 to 2001 as compared to the NYSDEC chronic water quality standard. The standard for unionized ammonia (as NH_3) is dependent on pH and temperature for different classes and specifications (NYSDEC, 1998). The state standard was consistently exceeded in the epilimnion and hypolimnion.

Concentrations of nitrite in Onondaga Lake are shown in Figure 8-12 for all epilimnion depths (0 to 9 m). Data from 2000 and the first half of 2001 indicate an improvement in water quality during the first half of the year, with exceedances of the NYSDEC standard of 0.1 mg/L for warm-water fish seen later in the year (i.e., summer and fall).

Figure 8-13 presents concentrations of phosphorus in Onondaga Lake in 1992. The most noticeable trend was the increasing concentrations of phosphorus in the hypolimnion during the period of stratification. The decrease in total phosphorous during the summer in the epilimnion is consistent with Effler et al. (1996). The NYSDEC guidance value of 20 μg/L (NYSDEC, 1998) was exceeded in the hypolimnion during the summer months in 1992. From 1997 to 2001, phosphorus concentrations were also elevated in the epilimnion samples (3 m depth based on human exposure), with almost all measurements above the guidance value (Figure 8-14). However, exceedances of an aesthetic effects guidance value are considered to have minimal impact on fish and wildlife in the area.

Tributary Water and Metro Discharge

The only nutrient form of nitrogen or phosphorus that was measured in tributaries and point sources in 1992 was ammonia (phosphorus was not measured). Mean concentrations of ammonia in the tributaries and Metro effluent under the three flow conditions are depicted in Figure 8-15.

Under base-flow conditions, maximum ammonia concentrations were highest in the Metro effluent (14 mg/L) followed by East Flume (4.2 mg/L), the lake outlet (2.5 mg/L), and Geddes Brook (1.6 mg/L), and were less than 1 mg/L in the remaining tributaries. Under intermediate flow conditions, maximum ammonia concentrations were highest in the Metro effluent (22 mg/L), followed by the East Flume (5.1 mg/L) and the lake outlet (3.0 mg/L), and were less than 3 mg/L elsewhere. Under high-flow conditions, maximum ammonia concentrations were also highest in the Metro effluent (15 mg/L), followed by the East Flume (5.0 mg/L) and the lake outlet (3.0 mg/L), and were 2 mg/L or less in the remaining tributaries.

8.1.1.9 Sulfide

The concentrations of sulfide in the hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-16. At depths of both 14 and 18 m, sulfide concentrations became measurable in early June to mid-July with the onset of stratification, increased substantially throughout the summer to maximum concentrations in late September, and then declined at 18 m after fall turnover. Sulfide concentrations were also elevated in the hypolimnion from 1997 to 2001 (Figure 8-17), but only depths of 12 to 18 m were sampled during this period.

8.1.1.10 Dissolved Oxygen

Lake Water

The concentrations of DO in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-18 and are indicative of a highly eutrophic lake. The ionic waste discharge of the chlor-alkali facility exacerbated the lake's problem of limited oxygen resources through alteration of the system's density stratification/mixing regime (Effler et al., 1996). Density differences have been reduced, but not eliminated, following closure of the facility (Owens and Effler, 1996). Prolonged stratification extends the period of anoxia in the lake.

The depth of anoxia (no oxygen) is presented from April to November 1992 in Figure 8-18. In the epilimnion, DO concentrations declined from 14 mg/L in April to 6 mg/L in early October. In the hypolimnion, DO concentrations rapidly declined from 13 mg/L in April to less than 0.5 mg/L in mid-June after the onset of stratification. Anoxic conditions remained in the hypolimnion until fall turnover. The depth of anoxia became established at 18 m in early June with the onset of stratification, rose to a depth of 9 to 12 m during the period of stratification, and then disappeared after fall turnover.

Dissolved oxygen concentrations sampled in 1997 to 2001 by Onondaga County have regularly failed to meet the NYSDEC standard of 4 mg/L in Onondaga Lake (Figure 8-20). Low DO levels were seen primarily at deeper depths. The failure to meet the DO standard only occurred at depths of 3 m or less in the northern basin during late October 1997 and 1998, around the time of fall turnover. The DO standard was always met at depths less than 3 m in the southern basin. Samples from later than October are not available for years other than 1997 and 1998.

Tributary Water and Metro Discharge

Mean DO concentrations in the tributaries and Metro effluent are presented for base-flow, intermediate-flow, and high-flow conditions in Figure 8-19. Under base-flow conditions, mean DO concentrations were lowest in the East Flume (5.6 mg/L) and were higher than 7.2 mg/L in the remaining tributaries. Under intermediate-flow conditions, mean DO in the East Flume was 5.9 mg/L and was higher than 7.4 mg/L elsewhere. Under high-flow conditions, mean DO concentrations were again lowest at the East Flume (5.7 mg/L) and were higher than 8.6 mg/L in the other tributaries sampled.

8.1.1.11 Water Transparency

The Secchi disk depths measuring visibility in Onondaga Lake from April to November 1992 are presented in Figure 8-21. Secchi disk depth generally varied from 1 to 2 m between April and early October and then increased to almost 6 m by the end of November, following fall turnover.

8.1.2 Distribution of Chemicals and Stressors of Concern in Onondaga Lake Surface Sediments

In this section, the distribution of COCs/SOCs in the surface sediments of Onondaga Lake is described. Detailed summaries of COC/SOC concentrations in sediments and exceedances of sediment quality values are presented in Appendices E (NYSDEC screening values) and F (site-specific sediment effect concentrations [SECs] calculated by TAMS) of this report. Chapter 5 of the RI (TAMS, 2002b) provides a detailed characterization of lake sediment contamination (surface and subsurface).

The 1992 lake sediment sampling provided a characterization of contamination throughout the entire lake, whereas the 2000 (Phase 2A) sampling focused on areas with higher levels of contamination, such as the southwestern portion of the lake and the mouth of Ninemile Creek. Surface sediments collected in 1992 were collected from the top 2 cm of the sediment column; whereas in 2000, surface samples were collected from the top 15 cm of sediment. The 0 to 15 cm zone of surficial sediment represents a fuller range of potential exposure of macroinvertebrates (Larson, pers. comm., 1999a). Benthic macroinvertebrates, and in particular oligochaetes, are commonly known to turn over as much as the top 20 cm of sediment in lakes, re-exposing contaminated sediments or concentrating the contaminants from sediments and making them available to higher level trophic receptors.

8.1.2.1 Mercury

The concentrations of total mercury in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figure 8-22. Mercury concentrations were elevated throughout most of the lake, with maximum and mean detected concentrations in 1992 of 69 mg/kg and 4.0 mg/kg. Most samples had concentrations of mercury greater than 2.0 mg/kg, with the highest concentrations of mercury (>10 mg/kg) found in the southwest corner of the lake between the East Flume and the Metro outfall. Mercury concentrations throughout most of the nearshore zone were less than 2.0 mg/kg, whereas concentrations in most of the deeper parts of the lake were between 2.0 and 5.0 mg/kg. Both maximum and mean values were above all sediment screening values (Chapter 4, Table 4-5).

The 2000 sampling effort focused on the southwestern portion of the lake and the area near the mouth of Ninemile Creek (Figure 8-22). The maximum concentration of 78 mg/kg was detected offshore from the East Flume outlet. In 2000, the highest levels of mercury were found between the East Flume and Harbor Brook, with high levels also detected near the mouth of Ninemile Creek. The mean detected concentration of the surface sediment samples was 8.0 mg/kg (compared to the 1992 mean of 4.0 mg/kg), which may result from the more directed sampling locations and the greater depth of the samples. Most samples collected in 1992 and 2000 exceeded NYSDEC sediment screening values (Appendix E), with the majority of samples and the means exceeding the NYSDEC severe effect level (SEL) of 1.3 mg/kg.

Mercury associated with sediment particles may be available to the food chain for the following reasons:

- Particle-borne mercury serves to maintain dissolved-phase concentrations of mercury, both in the water column of the lake as well as in porewater in the sediments.

- Sediment mercury represents an important pathway for ecological exposure via benthic invertebrate activity on the lake bottom.

- Water column mercury, as maintained by the suspended particle load as well as by sediment resuspension, represents an alternate exposure route for fish via direct uptake through the gills.

- Resuspension may not be the only way in which the mercury reenters the water column. Dissolution at the sediment/water interface and porewater migration can also serve to release sediment-bound mercury to the water column.

8.1.2.2 Other Metals

The concentrations of metals other than mercury in the surface sediments of Onondaga Lake (0 to 2 cm for 1992 data and 0 to 15 cm for 2000 data) are presented in Figures 8-23 through 8-29 (figures are not provided for metals with limited data; e.g., antimony). The following distributions were found for each metal in surface sediments:

- **Antimony** – In 1992, antimony was analyzed in 19 sediment samples at the 0 to 2 cm interval. Detections ranged between 3.1 and 6.4 mg/kg, with a mean of 4.6 mg/kg. In 2000, antimony was detected in 44 of 85 sediment samples at the 0 to 15 cm interval, ranging between 0.3 and 5.4 mg/kg. Of these samples, all of the 1992 and eight of the 44 in 2000 exceeded the NYSDEC LEL of 2 mg/kg.

- **Arsenic** (Figure 8-23) – In 1992, the highest concentration of arsenic (11 mg/kg) was detected near the mouth of Tributary 5A. The mean lakewide concentration was 3 mg/kg. The 2000 sampling, concentrated along the southwestern shore of the lake, detected a maximum concentration of 47 mg/kg and a mean concentration of 6.1 mg/kg. The highest arsenic concentrations were identified near the Interstate 690 (I-690) storm drainage discharge location. Maximum values exceeded screening values, as did the 2000 mean.

- **Cadmium** (Figure 8-24) – Concentrations in 1992 throughout most of the nearshore zone of the lake were less than 2 mg/kg, whereas concentrations in most of the deeper parts of the lake were between 2 and 5 mg/kg. The highest concentrations of cadmium were found off Harbor Brook and Ley Creek and at several stations along the eastern and western shorelines. The maximum and mean concentrations of cadmium detected in the lake in 1992 were 14 and 2.5 mg/kg, respectively. In 2000, the maximum and mean concentrations detected were 15 and 2 mg/kg, with the highest values detected near the mouths of Ley Creek and the East Flume. Mean and maximum concentrations were above some of the sediment screening values.

- **Chromium** (Figure 8-25) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg. The highest concentrations of chromium (up to 1,990 mg/kg) were found off Harbor Brook, between Tributary 5A and the East Flume, and at stations off the western shoreline near Wastebeds 1 through 8. The mean 1992 concentration of chromium detected in the lake was 81 mg/kg. In 2000, more intensive sampling along the southwestern shore of the lake confirmed that the area between Tributary 5A and the East Flume had the highest concentrations of chromium in surface sediments of the lake (up to 4,180 mg/kg). The mean concentration of the 2000 samples was 225 mg/kg. Both maximum and mean concentrations for both sampling periods were above sediment screening values.

- **Copper** (Figure 8-26) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg. The highest concentrations of copper (up to 173 mg/kg) were found off Harbor Brook and Tributary 5A and the lakewide mean concentration was 44 mg/kg. The maximum concentration detected in 2000 was 366 mg/kg, with a mean concentration of 66 mg/kg. The highest concentrations were detected between Tributary 5A and the East Flume. Maximum and mean concentrations were above sediment screening values for both sampling periods.

- **Lead** (Figure 8-27) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg, yielding an overall mean concentration of 51 mg/kg in the lake. The highest concentrations of lead (up to 251 mg/kg) were along the shoreline between Tributary 5A and Ley Creek. In 2000, more focused sampling confirmed that the areas with the highest levels of contamination were located between Tributary 5A and Ley Creek. The maximum concentration of 750 mg/kg was located near the mouth of Harbor Brook, where it may potentially enter Wetland SYW-19. The mean concentration detected in 2000 was 93 mg/kg. Both maximum and mean concentrations exceeded screening values.

- **Manganese** – In 1992, manganese was detected in all 19 sediment samples at the 0 to 2 cm interval and ranged between 93 and 508 mg/kg, with a mean of 278 mg/kg. In 2000, manganese was detected in all 85 sediment samples at the 0 to 15 cm interval, ranging between 107 and 1,190 mg/kg, with a mean of 334 mg/kg. Of these samples, only one of the 1992 and 11of the 85 in 2000 exceeded the NYSDEC LEL of 460 mg/kg. All of these exceedances were located near the I-690 lakeshore area.

- **Nickel** (Figure 8-28) – Concentrations in 1992 throughout the nearshore zone of the lake were generally less than 20 mg/kg, whereas concentrations throughout the deeper parts of the lake generally were between 20 and 50 mg/kg. The highest concentrations of nickel (up to 650 mg/kg) were found between Tributary 5A and the East Flume. The mean concentration detected in 1992 was 28 mg/kg. The 2000 sampling showed a maximum concentration of 1,670 mg/kg, detected near the mouth of Tributary 5A, and a mean concentration of 84 mg/kg. Maximum and mean concentrations during both periods exceeded screening values.

- **Selenium** – In 1992, selenium was detected in 12 of the19 sediment samples at the 0 to 2 cm interval and ranged between 0.3 and 1.1 mg/kg, with a mean of 0.5 mg/kg. In 2000, selenium was detected in 49 of 85 sediment samples at the 0 to 15 cm interval ranging between 0.5 and 5.9 mg/kg, with a mean of 1.6 mg/kg. There are no selenium sediment screening criteria. This COC was retained based on it being a COC in fish.

- **Silver** – In 1992, silver was detected in14 of the19 sediment samples at the 0 to 2 cm interval and ranged between 0.6 and 5.1 mg/kg, with a mean of 1.5 mg/kg. In 2000, silver was detected in 53 of 85 of the 0 to 15 cm sediment samples analyzed, ranging between 0.1 and 6.1 mg/kg with a mean of 1.4 mg/kg. Of these detections, only seven of the 1992 and 20 of the 2000 samples exceeded the NYSDEC LEL of 1 mg/kg. In 1992 the exceedances were scattered throughout the lake, while in 2000 all of the exceedances were located in the southern basin.

- **Vanadium** – In 1992, vanadium was detected in 14 of the19 sediment samples at the 0 to 2 cm interval and ranged between 0.5 and 101 mg/kg, with a mean of 12.7 mg/kg. In 2000, silver was detected in all 85 of the 0 to 15 cm sediment samples analyzed ranging between 1.8 and 319 mg/kg, with a mean of 24 mg/kg. There are no sediment screening criteria. This COC was retained based on it being a COC in fish.

- **Zinc** (Figure 8-29) – Concentrations in 1992 throughout the northern nearshore zone of the lake generally were less than 120 mg/kg, whereas concentrations throughout the deeper areas of the lake were between 120 and 150 mg/kg in the northern part and between 150 and 270 mg/kg in the southern part of the lake. The highest concentrations of zinc (>200 mg/kg, maximum of 276 mg/kg) were found off Harbor Brook and Ley Creek, and at discrete stations off the western shoreline in the southern basin. In 2000, the maximum concentration of 421 mg/kg was detected nearshore between Tributary 5A and the East Flume. The mean concentration was 122 mg/kg. Maximum concentrations in 1992 and 2000 and the mean concentration in 1992 were above sediment screening values.

8.1.2.3 Benzene, Toluene, Ethylbenzene, and Xylenes Compounds

The concentrations of BTEX compounds in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-30 through 8-33. Concentrations of all four of these compounds were less than 50 µg/kg throughout much of the lake, however, elevated concentrations of all four compounds were found along the southwestern shoreline of the lake. The highest concentrations (>1,000 µg/kg) of all four compounds were detected off Harbor Brook and the East Flume, with increased concentrations stretching along the southern shoreline until north of Ley Creek. The maximum detected concentrations of benzene, toluene, ethylbenzene, and xylene were 5,700, 4,200, 1,300, and 13,000 µg/kg, respectively, and mean concentrations of these compounds were 440, 150, 660, and 3,400 µg/kg, respectively.

Higher concentrations of BTEX compounds were detected in 2000. The maximum detected concentrations of BTEX were 42,000, 8,300, 71,000, and 330,000 µg/kg, respectively, and mean concentrations of these compounds were 2,600, 1,900, 2,900, and 24,000 µg/kg, respectively.

Concentrations detected in both 1992 and 2000 exceeded sediment screening values. Most samples exceeding screening values were collected from the southwestern shoreline of the lake.

8.1.2.4 Chlorinated Benzenes

The concentrations of chlorinated benzenes (i.e., monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene) in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-34, 8-35, 8-36, and 8-37. Concentrations of all of these compounds in 1992 were less than 100 µg/kg throughout most of the lake. However, concentrations of all of these compounds were sharply elevated along the southwestern shoreline of the lake. The highest concentrations (>1,000 µg/kg) were found in the area between Harbor Brook and the I-690 outfalls, which are located at the approximate border of the Semet Residue Ponds and the Willis Avenue sites. Maximum detected concentrations of monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene in 1992 were 43,000, 22,800, 4,200, and 1,200 µg/kg, respectively, and mean concentrations of these compounds were 2,700, 2,400, 1,100, and 63 µg/kg, respectively.

Samples collected in 2000 exhibited higher concentrations of contaminants with maximum concentrations of monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene of 1,000,000, 239,000, 35,000, and 6,750 µg/kg, respectively, and mean concentrations of these compounds were 34,700, 16,500, 6,800, and 180 µg/kg, respectively. Concentrations detected in both 1992 and 2000 exceeded sediment screening values. Benthic acute screening criteria for monochlorobenzene and dichlorobenzenes were exceeded at several locations along the southwestern shore.

8.1.2.5 Total Polychlorinated Biphenyls

The concentrations of total PCBs in the surface sediments of Onondaga Lake in 1992 (0 to 2 cm) and 2000 (0 to 15 cm) are presented in Figure 8-38. Concentrations of PCBs were less than 500 µg/kg throughout most of the lake, but even sediment above 10 µg/kg exceeds sediment screening values. Elevated levels of PCBs were concentrated in the southern part of the lake. The highest concentrations (>500 µg/kg) in 1992 were found primarily in the nearshore zone between Tributary 5A and Ley Creek. The maximum detected concentration in 1992 was 2,100 µg/kg, with a mean concentration of 290 µg/kg.

In 2000, the maximum detected concentration was 21,000 µg/kg, with a mean concentration of 1,100 µg/kg. The maximum concentration was detected slightly offshore from the mouth of the East Flume. Mean and maximum concentrations of PCBs from both sampling events exceeded sediment screening values.

8.1.2.6 Polycyclic Aromatic Hydrocarbon Compounds

PAHs were divided into LPAHs (low molecular weight PAHs: fluorene, naphthalene and 2-methylnaphthalene) and HPAHs (high molecular weight PAHs: acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, fluoranthene, indeno[1,2,3-cd]pyrene, phenanthrene, and pyrene), based on results of the principal component analysis (PCA) presented in the RI (see Appendix I and Chapter 6 of the RI) (TAMS, 2002b).

The concentrations of PAH compounds in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-39 and 8-40. Concentrations of these compounds were less than 1,000 µg/kg throughout most of the lake. However, elevated concentrations of these compounds were found in the southern part of the lake. The highest concentrations of both LPAHs (>5,000 µg/kg) and HPAHs (>10,000 µg/kg) were found primarily in the nearshore zone between Tributary 5A and Ley Creek. The 2000 sampling also revealed elevated levels of LPAHs (>5,000 µg/kg) between Tributary 5A and Harbor Brook and along Wastebeds 1 through 8. As is true for LPAHs, the HPAHs are found at high concentrations (>10,000 µg/kg) throughout the southern basin. The highest concentrations occur off the Oil City shoreline region, as well as the Honeywell shoreline area. From 2 to 6 m, both areas remain as centers of contamination. Individual and total PAHs exceeded sediment screening values in 1992 and 2000. The maximum surface concentration of naphthalene (26,000,000 µg/kg at 0 to 5 cm) was detected at Station S435 near the Semet Residue Ponds site.

8.1.2.7 Dioxins and Furans

Dioxins and furans were analyzed in lake sediments in 2000 and are presented in Figure 8-41 as avian toxicity equivalence quotients (TEQs), which are more conservative than the corresponding mammalian TEQs. The maximum avian TEQ in surface sediments (0 to15 cm) of 524 ng/kg was found at Station S346 near the East Flume. The mean and minimum avian TEQs were 117 and 4.7 ng/kg, respectively. The maximum mammalian TEQ was also found at Station S346 at 165 ng/kg, and the mean and minimum mammalian TEQs were 44 and 1.4 ng/kg, respectively.

8.1.2.8 Other Sediment Chemicals of Concern

Six other organic compounds/groups of compounds were selected as sediment COCs for this BERA. The concentrations of these compounds in surface sediments are shown in Figures 8-42 to 8-47. The following distributions were found for each COC:

- **Phenol** (Figure 8-42) – In 1992, phenol was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in two samples at a concentration of 45 μg/kg. In 2000, phenol was analyzed for in 84 surface samples and detected in 11 samples with a range of concentrations from 190 to 2,600 μg/kg. The maximum sediment concentration was found at Station S349 near the East Flume.

- **Dibenzofuran** (Figure 8-43) – In 1992, dibenzofuran was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in five of those samples, with concentrations ranging from 18 to 1,300 μg/kg. In 2000, dibenzofuran was analyzed in 84 samples and detected in 18 samples, with a maximum concentration of 81,000 μg/kg at Station S313 near Harbor Brook.

- **DDT and Metabolites** (Figure 8-44) – In 1992, DDT and its metabolites were analyzed in 19 sediment samples from the 0 to 2 cm depth interval and were detected in four of those samples. The range of detections was from 7.4 to 47 μg/kg. In 2000, DDT and its metabolites were analyzed in 84 samples and detected in 54 samples, with a range of concentrations from 1.1 to 88 μg/kg. The maximum sediment concentration of DDT in the 0 to15 cm depth interval was found at Station S313 near Harbor Brook.

- **Chlordane** (Figure 8-45) – In 1992, chlordane was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in one sample at a concentration of 5.1 μg/kg at Station S1 near the mouth of Harbor Brook. In 2000, chlordane was analyzed in 84 samples and detected in 26 samples, with a range of concentrations from 1.1 to 50 μg/kg. The maximum sediment concentration of chlordane in the 0 to 15 cm depth interval was found at Station S314 near Harbor Brook.

- **Dieldrin** (Figure 8-46) – In 1992, dieldrin was not detected in surface sediment samples. In 2000, dieldrin was analyzed in 84 samples and detected in 26 samples, with concentrations ranging from 1.3 to 36 μg/kg. The maximum sediment concentration of dieldrin in the 0 to 15 cm depth interval was found at Station S338 near Tributary 5A.

- **Heptachlor and Heptachlor Epoxide** (Figure 8-47) – In 1992, heptachlor and heptachlor epoxide were analyzed in 19 sediment samples from the 0 to 2 cm depth interval and detected in none of those samples. In 2000, heptachlor and heptachlor epoxide were analyzed in 84 samples and detected in 28 samples, with a range of concentrations from 0.8 to 52 μg/kg. The maximum sediment concentration of heptachlor and heptachlor epoxide in the 0 to 15 cm depth interval was found at Station S314 near Harbor Brook.

8.1.2.9 Calcium and Oncolites

The concentrations of calcium carbonate in the surface sediments of Onondaga Lake in 1992 are presented in Figure 8-48. Quantities of oncolites were estimated by determining the volume (in mL) of oncolites retained by a sieve with mesh size of 2 mm (i.e., gravel size) for the sieved fraction of all 0.06-m^2 benthic macroinvertebrate samples. No oncolite survey was performed in 2000. The following distributions were found:

- **Calcium** – Concentrations of calcium carbonate were generally greater than 60 percent throughout most of the nearshore zone and were generally less than 60 percent throughout most of the deeper parts of the lake. The highest calcium carbonate concentrations (>80 percent) were found in the nearshore zone off Ley Creek and Tributary 5A, along much of the northwestern and northeastern shorelines, and at several stations off the eastern and western shorelines.

- **Oncolites** – The distribution of oncolites was determined only for the nearshore zone because they are not found in the deeper parts of the lake (possibly because the degree of calcite oversaturation is greater in the epilimnion than the hypolimnion). In general, the distribution of oncolites closely corresponded to the distribution of calcium carbonate, of which they are composed. The lowest concentrations (<100 mL/0.06 sq m) were found between Tributary 5A and Ley Creek and along sections of the eastern and western shorelines. The highest concentrations (>300 mL/0.06 sq m) were found along most of the northwestern and northeastern shorelines and in small areas off Ley Creek and Tributary 5A.

Sandy material was mixed in with the oncolites in much of the high concentration zone, while silts and soft fine-grained sediments composed mostly of calcium carbonate occurred primarily on the southwestern shoreline and in the area just north of the wastebeds.

8.1.2.10 Sediment Porewater Characterization

Sediment cores for porewater were collected at four locations in 1992 and seven locations in 2000. The chloride profiles from the 1992 porewater samples obtained in August and November are strikingly different throughout the length of the cores. The August concentrations are consistently and substantially lower, which could indicate a rapid mechanism for movement through the sediments; however, it most likely indicates that the lake water and porewater were allowed to mix during collection of the August samples, which would invalidate the samples. Because of the significant change in chloride concentrations, the August and possibly all of the porewater results from 1992 are suspect and unusable and, therefore, were not used in this analysis.

The samples in 2000 were sectioned at 4 cm intervals for the top 8 cm and then one more 4 cm interval was collected at depths ranging from 30 to 120 cm, for a total of three intervals per core. These samples were analyzed for total mercury, methylmercury, and other analytes in porewater and solids. Four of the 2000 cores (from Stations S305, S344, S402, and S405) were from sediments above the thermocline, and three (from Stations S303, S354, and S355) were from below the thermocline. Three replicate cores were obtained at each station.

Mercury

Figures 8-49 through 8-55 show the distribution of total mercury and methylmercury in dissolved (filtered) form in porewater from the seven locations sampled in 2000 at Stations S303, S305, S344, S354, S355, S402, and S405, respectively. They also show the concentration in overlying water. In all cases, the porewater concentration in the 0 to 4 cm interval was higher than in the overlying water. The highest concentrations of total mercury and methylmercury occurred at Stations S344 and S402, located offshore of the East Flume within an area that contains Solvay waste material. There was no consistent pattern of concentration in porewater with depth probably because other parameters (e.g., porewater pH and sulfide concentration) influence mercury concentration independent of depth. There was also a poor correlation between concentration in porewater and concentration in the solids. Concentrations of mercury in sediments at these locations ranged from less than 1 to 66 mg/kg.

8.1.3 Distribution of Chemicals and Stressors of Concern in Wetland Soils/Sediments

The distribution of COCs in the wetland soils/sediments is discussed in this section. Sediment cores were collected in 2000 from four Onondaga Lake wetlands, including two wetlands located at the north end of the lake (Wetland SYW-6) and the mouth of Ninemile Creek (Wetland SYW-10), and two southern wetlands located at the mouth of Harbor Brook (Wetland SYW-19), and the mouth of Ley Creek

(Wetland SYW-12). Four stations were sampled in each wetland at two depth intervals (0 to 15 and 15 to 30 cm). In addition, SYW-6 was sampled by NYSDEC/TAMS in 2002 at five additional locations. Summary statistics for BERA COCs and screening against appropriate sediment criteria and ecological soil screening values are summarized in Appendix D of this BERA and figures can be found in Chapter 5 of the RI. In this BERA, only the 0 to15 cm interval is used in evaluating risks to receptors.

8.1.3.1 Mercury

Detections of total mercury in the 0 to 15 cm interval ranged from 0.05 to 25 mg/kg, with the maximum concentration occurring at Station S385 in Wetland SYW-19. Average total mercury concentrations in the surface sediment layers was 6.3 mg/kg. Total mercury concentrations in Wetland SYW-19 were significantly higher than values reported for the other wetlands. This wetland area is part of Honeywell's Wastebed B/Harbor Brook site and will be further evaluated as part of the RI/FS for that site. About 86 percent of the reported total mercury concentrations exceeded the LEL of 0.15 mg/kg, and 48 percent exceeded the SEL of 1.3 mg/kg. All samples exceeded the earthworm toxicity and USEPA Region 4 soil screening value of 0.1 mg/kg, except two samples (from Stations S387 and S390) located in Wetland SYW-12. About 76 percent of mercury concentrations exceeded the phytotoxicity screening value of 0.3 mg/kg and no samples exceeded the microbial toxicity screening value of 30 mg/kg.

8.1.3.2 Other Metals

Twelve metals in addition to mercury (i.e. arsenic, cadmium, chromium, copper, cyanide, lead, nickel, selenium, silver, thallium, vanadium, and zinc) were selected as plant COCs based on phytotoxicity (Efroymson et al., 1997a) (Chapter 6, Table 6-1). Additional metals (i.e., antimony, barium, iron, and manganese) based on microbial toxicity (Efroymson et al., 1997b), earthworm toxicity (Efroymson et al., 1997b), and USEPA Region 4 (1999) soil screening values were also selected as soil COCs (Table 6-1).

- **Antimony:** Concentrations ranged from 0.17 to 2.2 mg/kg. The maximum concentration at Station 3 in Wetland SYW-6, sampled in 2002, exceeded the NYSDEC LEL of 2 mg/kg. No station exceeded the NYSDEC SEL of 25 mg/kg. No station exceeded the USEPA Region 4 and phytotoxicity soil values of 3.5 mg/kg and 5 mg/kg, respectively.

- **Arsenic:** Concentrations ranged from 0.5 to 18 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10, which exceeded the NYSDEC LEL of 6 mg/kg, along with three other stations. No station exceeded the NYSDEC SEL of 33 mg/kg. The maximum concentration at Station S379 in Wetland SYW-10 also exceeded the phytotoxicity and USEPA Region 4 soil screening value of 10 mg/kg. No station exceeded the earthworm or the microbial toxicity values of 60 and 100 mg/kg, respectively.

- **Barium:** Concentrations ranged from 31.4 to 390 mg/kg. About 24 percent of the samples (5 of 21) exceeded the USEPA Region 4 soil screening value of 165 mg/kg. No station exceeded the phytotoxicity soil value of 500 mg/kg.

- **Cadmium:** Concentrations ranged from 0.14 to 14 mg/kg, with the maximum concentration occurring at Station S375 in Wetland SYW-6, which exceeded the NYSDEC SEL of 9 mg/kg. About 86 percent of cadmium concentrations exceeded the LEL of 0.6 mg/kg. About 48 percent of cadmium concentrations exceeded the USEPA Region 4 screening value of 1.6 mg/kg and about 19 percent exceeded the phytotoxicity screening value of 4 mg/kg. No station exceeded the earthworm or the microbial toxicity values of 20 mg/kg.

- **Chromium:** Chromium concentrations ranged from 11 to 154 mg/kg, with 19 stations exceeding the LEL of 26 mg/kg and two stations exceeding the SEL of 110 mg/kg. All stations exceeded the USEPA Region 4 and earthworm toxicity screening value of 0.4 mg/kg and the phytotoxicity and microbial toxicity values of 1 and 10 mg/kg, respectively.

- **Copper:** Concentrations ranged from 9.5 to 167 mg/kg, with 19 of 21 samples exceeding the LEL of 16 mg/kg and two samples exceeding the SEL of 110 mg/kg. About 52 percent of the samples (11 of 21) exceeded the USEPA Region 4 screening value of 40 mg/kg. About 38 percent (8 out of 21 samples) exceeded the earthworm toxicity value of 50 mg/kg and two samples exceeded the phytotoxicity and microbial toxicity value of 100 mg/kg.

- **Cyanide:** Cyanide had only one detection of 5.4 mg/kg, which occurred at Station S376 in Wetland SYW-6. This detection exceeded the USEPA Region 4 screening value of 5 mg/kg.

- **Iron:** Concentrations ranged from 3,290 to 24,000 mg/kg. Two samples (Station S379 in Wetland SYW-10 and Station 2 in SYW-6) exceeded the NYSDEC LEL of 20,000 mg/kg. No station exceeded the NYSDEC SEL of 40,000 mg/kg. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 200 mg/kg.

- **Lead:** Concentrations ranged from 18 to 259 mg/kg. The maximum concentration was found at Station S385 in Wetland SYW-19. The lead LEL of 31 mg/kg and the SEL of 110 mg/kg were exceeded in 15 and 6 samples out of 21 samples, respectively. About 62 percent of the samples (13 of 21) exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 900 and 500 mg/kg, respectively.

- **Manganese:** Concentrations ranged from 163 to 488 mg/kg. The maximum concentration, detected at Station S381 in Wetland SYW-10, exceeded the NYSDEC LEL of 460 mg/kg. No station exceeded the NYSDEC SEL of 1,100 mg/kg. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 100 mg/kg, while no detected concentration exceeded the phytotoxicity soil value of 500 mg/kg.

- **Nickel:** Concentrations ranged from 5.5 to 64 mg/kg. The maximum concentration was found at Station S375 in Wetland SYW-6, which exceeded both the LEL of 16 mg/kg and the SEL of 50 mg/kg. A majority of stations exceeded the LEL. Fourteen sediment samples exceeded the NYSDEC LEL and one sample exceeded the SEL. About 33 percent of the samples (7 of 21) exceeded the USEPA Region 4 and phytotoxicity soil value of 30 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 90 and 200 mg/kg, respectively.

- **Selenium:** Concentrations ranged from 0.9 to 2.5 mg/kg, with the maximum concentration occurring at Station S375 in Wetland SYW-6. Only soil screening values were available for selenium. All detected concentrations exceeded the USEPA Region 4 screening value of 0.81 mg/kg. About 86 percent, or six out of the seven detected concentrations, exceeded the phytotoxicity value of 1 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 100 and 70 mg/kg, respectively.

- **Silver:** Concentrations ranged from 0.2 to 2.7 mg/kg, with the maximum concentration occurring at Station S389 in SYW-12, which exceeded both the LEL of 1.1 mg/kg and the SEL of 2.2 mg/kg. Four samples exceeded the NYSDEC LEL and only one sample exceeded the SEL. The maximum concentration also exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg, but not the microbial toxicity value of 50 mg/kg.

- **Thallium:** Concentrations ranged from 1 to 2.5 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10. Only soil screening values were available for thallium. Two of three detected concentrations exceeded the USEPA Region 4 and phytotoxicity soil value of 1 mg/kg.

- **Vanadium:** Concentrations ranged from 3.4 to 30.6 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10. Only soil screening values were available for vanadium. All samples exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg and two samples exceeded the microbial toxicity value of 20 mg/kg.

- **Zinc:** Concentrations of zinc ranged from 34 to 510 mg/kg, with the maximum concentration found at Station S376 in Wetland SYW-6. Concentrations exceeded the LEL of 120 mg/kg and SEL of 270 mg/kg in eight and two samples, respectively. All but one sample exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg. About 67 percent of the samples (14 of 21) exceeded the microbial toxicity value of 100 mg/kg and about 19 percent of the samples (four of 21) exceeded the earthworm toxicity value of 200 mg/kg.

8.1.3.3 Organic Contaminants

Thirteen organic COCs/COC groups (i.e., benzene, chlorobenzene, dichlorobenzenes [sum], trichlorobenzenes [sum], hexachlorobenzene, phenol, total PAHs, aldrin, chlordanes, DDT and metabolites, dieldrin, hexachlorocyclohexanes, and total PCBs) were selected in soils. Plant screening values were not available for organic compounds. Summary statistics of wetland soils/sediments are presented in Appendix H, Tables H-15 to H-19.

- **Benzene:** Concentrations of benzene ranging from 3.9 to 60 µg/kg were detected in three of 12 wetland samples. The maximum concentration was found at Station S384 in Wetland SYW-19. Only this sample exceeded the NYSDEC chronic benthic sediment criterion of 28 µg/gOC when normalized by its respective organic carbon content. This sample also exceeded the USEPA Region 4 soil screening value of 50 µg/kg dry weight.

- **Chlorobenzene** (i.e., monochlorobenzene): Concentrations of chlorobenzene ranging from 2 to 600 µg/kg were found in the southern wetlands. Chlorobenzene was detected in five of 12 samples, with the maximum concentration found at Station S384 in Wetland SYW-19. Only three of the five stations exceeded the NYSDEC chronic benthic sediment criterion of 3.5 µg/gOC when normalized by the sample-specific organic carbon content. These three samples also exceeded the USEPA Region 4 soil screening value of 50 µg/kg dry weight. No station exceeded the earthworm toxicity value of 40,000 µg/kg dry weight.

- **Dichlorobenzenes:** Elevated levels of dichlorobenzenes (sum) were found in the wetlands around Onondaga Lake. Concentrations ranged from 54 to 14,700 µg/kg, with the highest concentrations in the four stations in Wetland SYW-19. When the concentrations of the sum of dichlorobenzenes are normalized by their respective organic carbon content, all values in Wetland SYW-19 exceeded the NYSDEC chronic benthic criterion of 12 µg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 10 µg/kg dry weight.

- **Trichlorobenzenes:** Concentrations of 1,2,4,-trichlorobenzene ranging from 200 to 6,600 µg/kg were found in the wetlands around Onondaga Lake, with the highest concentration at Wetland SYW-19. 1,2,4,-Trichlorobenzene was detected in five of 21 samples collected. Only four of the five stations exceeded the NYSDEC chronic benthic criterion of 91 μg/gOC for total trichlorobenzenes when normalized by organic carbon content. All detected concentrations exceeded the USEPA Region 4 soil screening value of 10 μg/kg dry weight.

- **Hexachlorobenzene:** Concentrations of hexachlorobenzene ranging from 2.7 to 5,400 μg/kg were found in the wetlands around Onondaga Lake, with the highest concentration at Wetland SYW-19. Hexachlorobenzene was detected in 12 of 21 samples, with four of the 12 exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 12 μg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 2.5 μg/kg dry weight, but none exceeded the microbial toxicity value of 1,000 mg/kg.

- **Phenol:** Concentrations of phenol ranging from 89 to 2,800 μg/kg were found in the wetlands around Onondaga Lake, with the highest concentration at Wetland SYW-19. Phenol was detected in three of 21 samples, with all three detections exceeding the NYSDEC chronic benthic sediment criterion of 0.5 μg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 50 μg/kg dry weight, but no station exceeded the phytotoxicity, microbial toxicity or earthworm toxicity values of 70, 100, and 30 mg/kg, respectively.

- **Polycyclic Aromatic Hydrocarbons:** Maximum concentrations for total PAHs occurred at Stations S384 and S385 in Wetland SYW-19. The organic carbon-normalized concentrations for benz(a)anthracene indicate that ten samples exceeded the NYSDEC chronic benthic sediment criterion of 12 μg/gOC. Total PAHs were detected in 19 of 21 samples, with all but two detections exceeding the USEPA Region 4 soil screening value of 1,000 μg/kg dry weight.

- **Aldrin:** Concentrations of aldrin detected in the wetlands ranged from 21 to 45 μg/kg. Aldrin was detected only in Wetland SYW-19, where it was found in three of four samples. All three detections exceeded the Ontario Ministry of the Environment (OME) LEL of 2 μg/kg and the NYSDEC wildlife bioaccumulation sediment criterion. These three detections also exceeded the USEPA Region 4 soil screening value of 2.5 μg/kg.

- **Chlordanes**: Concentrations of chlordanes ranged from 2.3 to 30 μg/kg, with the maximum concentrations detected in Wetland SYW-19. Chlordane was detected in six of 16 samples, with all detections exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 0.006 μg/gOC. There are no soil screening values for chlordane.

- **Dieldrin:** Concentrations of dieldrin ranged from 2.6 to 24 µg/kg, with elevated values detected in Wetlands SYW-19 and SYW-12. Dieldrin was detected in seven of 16 samples, with two detections exceeding the NYSDEC chronic benthic sediment criterion of 9 µg/gOC. All seven detected concentrations exceeded the USEPA Region 4 soil screening value of 0.5 µg/kg dry weight.

- **DDT and metabolites**: Concentrations of DDT and metabolites ranged from 1.2 to 51 µg/kg, with the maximum concentrations detected in Wetland SYW-19. DDT and metabolites were detected in 14 of 16 samples with exceedances of the NYSDEC chronic benthic sediment criterion of 1 µg/gOC. About 64 percent of the samples (nine of 14 detections) exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg dry weight.

- **Hexachlorocyclohexanes:** Concentrations of hexachlorocyclohexanes ranged from 1.7 to 10 µg/kg, with the maximum concentrations detected in Wetland SYW-19. Hexachlorocyclohexanes were detected in five of 16 samples, with all exceeding the USEPA sediment quality benchmark of 0.37 µg/gOC. All detected samples also exceeded the USEPA Region 4 soil screening value of 1 µg/kg dry weight.

- **Polychlorinated Biphenyls:** Concentrations in the wetlands around Onondaga Lake were highest in Wetlands SYW-19 and SYW-12, where Aroclors 1242, 1254, and 1260 were detected. Total PCBs ranged between 28 and 1,100 µg/kg. PCBs were detected in all samples analyzed, with all detections exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 1.4 µg/gOC. All detected values also exceeded the USEPA Region 4 soil screening value of 20 µg/kg dry weight. No station exceeded the phytotoxicity value of 40 mg/kg.

8.1.4 Distribution of Chemicals and Stressors of Concern in the Dredge Spoils Area

This section discusses the distribution of COCs in the dredge spoils area. Summary statistics for the dredge spoils area data are presented in Appendix D, Table D-50 and Appendix H, Table H-20, and further information, including figures, can be found in Appendix G1 of the RI (TAMS, 2002b). Surface and subsurface soil samples were collected by Honeywell in 2000 from dredged material disposal basins (Basins 1 to 4) located north of the mouth of Ninemile Creek, along the shoreline, in order to aid in characterizing the nature and extent of contamination of the dredged material and the fill placed on top of the spoils.

With the possible exception of Basin 4, these basins contain material dredged from the Ninemile Creek delta from 1966 to 1968. Eight sampling stations in the four basin areas were sampled at surface (0 to 60 cm) and subsurface (> 60 cm) intervals. The highest concentrations of many contaminants were found below 60 cm, making ecological receptor contact with these dredge spoils unlikely.

8.1.4.1 Mercury

Total mercury was detected at elevated concentrations within the dredge spoils area. Detections of total mercury in the top 60 cm ranged from 0.05 to 4 mg/kg, with higher concentrations (up to about 100 mg/kg) occurring in Basins 1 to 3 at subsurface intervals, confirming that contaminated sediments from the lake had been disposed of in this area. About 57 percent of the detected values (four of seven) exceeded the USEPA Region 4 and earthworm toxicity soil value of 0.1 mg/kg. Two detected values exceeded the phytotoxicity value of 0.3 mg/kg and none exceeded the microbial toxicity value of 30 mg/kg.

8.1.4.2 Other Metals

Some of the other metal COCs found in the dredge spoils soils are discussed in this section. Only soil values were used to screen the dredge spoils data (Chapter 5, Table 5-6), as the dredge spoils area is not regularly inundated by water.

- **Antimony:** Antimony concentrations ranged from 0.24 to 1.3 mg/kg in the 0 to 60 cm interval. The highest concentrations were seen in deeper samples not considered in this assessment. All detected values exceeded the USEPA Region 4 and phytotoxicity soil values of 3.5 and 5 mg/kg, respectively.

- **Arsenic:** Arsenic concentrations ranged from 3.2 to 8.4 mg/kg in the 0 to 60 cm interval. Higher concentrations were detected in deeper samples not considered in this assessment. No station exceeded the USEPA Region 4 and phytotoxicity soil value of 10 mg/kg.

- **Barium:** Barium concentrations ranged from 66.6 to 78.2 mg/kg in the 0 to 60 cm interval. Higher concentrations were seen in deeper samples not considered in this assessment. No station exceeded the USEPA Region 4 and phytotoxicity soil values of 165 and 500 mg/kg, respectively.

- **Cadmium:** In the top 60 cm, cadmium was undetected in all samples. In subsurface samples, concentrations ranged from 0.07 to 4.3 mg/kg. The highest concentration occurred at a depth of 230 to 250 cm in Basin 2, which is deeper than the surface soils considered in this assessment. However, the maximum detected concentration of 4.3 mg/kg exceeds both the USEPA Region 4 and phytotoxicity soil values of 1.6 and 4 mg/kg, respectively.

- **Chromium:** Chromium concentrations ranged from 12 to 29 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and earthworm toxicity soil value of 0.4 mg/kg, the phytotoxicity soil value of 1 mg/kg, and the microbial toxicity soil value of 10 mg/kg.

- **Copper:** Copper concentrations ranged from 11 to 26 mg/kg in the 0 to 60 cm interval. No station exceeded the USEPA Region 4 or earthworm toxicity soil values of 40 and 50 mg/kg, respectively. The highest concentrations were seen in deeper samples from Basin 3 that were not considered in this assessment.

- **Cyanide:** In the top 60 cm, cyanide was undetected in all samples. In deeper samples, concentrations ranged from 0.9 to 1.4 mg/kg, with higher subsurface concentrations in Basin 3. No station exceeded the USEPA Region 4 soil screening value of 5 mg/kg.

- **Iron:** Iron concentrations ranged from 9,260 to 15,900 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 200 mg/kg.

- **Lead:** Concentrations ranged from 4 to 14 mg/kg in the 0 to 60 cm interval, with elevated concentrations below 220 cm at Basin 2. No detected concentrations exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg.

- **Manganese:** Manganese concentrations ranged from 246 to 354 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 100 mg/kg, but none exceeded the phytotoxicity soil value of 500 mg/kg.

- **Nickel:** Nickel concentrations ranged from 9.3 to 13.8 mg/kg in the 0 to 60 cm interval. No detected concentration exceeded the USEPA Region 4 and phytotoxicity soil value of 30 mg/kg.

- **Selenium:** Concentrations ranged from 0.5 to 1.4 mg/kg. The highest surface soil concentration was detected in the 6 to 18 cm interval. About 75 percent of the samples (six of eight) exceeded the USEPA Region 4 soil screening value of 0.81 mg/kg. Only three detected samples exceeded the phytotoxicity soil value of 1 mg/kg. No detected samples exceeded the microbial toxicity or earthworm toxicity values of 100 and 70 mg/kg, respectively.

- **Silver:** In the top 60 cm, silver was undetected in all samples. In subsurface samples, concentrations ranged from 0.12 to 0.88 mg/kg. The highest concentration occurred at a depth of 219 to 244 cm, which is deeper than the surface soils considered in this assessment. However, the highest concentration did not exceed the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg.

- **Thallium:** In the top 60 cm, thallium was undetected in all samples. Thallium was detected only in the subsurface interval of 60 to 120 cm at Basin 3, with a value of 0.8 mg/kg. This concentration did not exceed the USEPA Region 4 and phytotoxicity soil value of 1 mg/kg.

- **Vanadium:** In the top 60 cm, vanadium concentrations ranged from 13.2 to 28.6 mg/kg, with the higher concentration occurring deeper than the surface soils considered in this assessment. All detected samples in the top 60 cm exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg. Only two samples exceeded the microbial toxicity soil value of 20 mg/kg.

- **Zinc:** In the top 60 cm, zinc concentrations ranged from 29 to 50 mg/kg, with the highest values occurring below 200 cm at Basin 2. One sample exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg, and none exceeded the microbial toxicity and earthworm toxicity soil values of 100 and 200 mg/kg, respectively.

8.1.4.3 Organic Contaminants

Concentrations of organic COCs detected in dredge spoils were as follows (note that VOCs, including chlorobenzene and BTEX compounds, were not analyzed in these dredge spoils samples):

- **Dichlorobenzenes:** In the 0 to 60 cm interval, dichlorobenzenes had only one detection, at 51 µg/kg for 1,2-dichlorobenzene. This concentration exceeded the USEPA Region 4 soil screening value of 10 µg/kg.

- **Hexachlorobenzene:** In the 0 to 60 cm interval, hexachlorobenzene was only detected once, at a concentration of 410 µg/kg. Maximum concentrations were found in the 60 to 150 cm interval at Station S438 in Basin 2. The detected sample in the 0 to 60 cm interval exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg.

- **Polycyclic Aromatic Hydrocarbons:** In the 0 to 60 cm interval, total PAH concentrations ranged from 38 to 1,541 µg/kg. The maximum detected sample exceeded the USEPA Region 4 soil screening value of 1,000 µg/kg. The highest concentration of total PAHs was 208,000 µg/kg, detected in the 180 to 210 cm interval in Basin 4. All the individual PAH COCs had at least two values exceeding the NYSDEC allowable soil concentration.

- **Aldrin:** Aldrin was analyzed in one sample at the 61 to 149 cm range at Station S438 and detected at a concentration of 1.2 µg/kg. This concentration did not exceed the USEPA Region 4 soil screening value of 2.5 µg/kg.

- **Dieldrin:** Dieldrin was analyzed in one sample at the 61 to 149 cm range at Station S438 and detected at a concentration of 3.8 µg/kg. This concentration exceeded the USEPA Region 4 soil screening value of 0.5 µg/kg.

- **Polychlorinated Biphenyls:** In the 0 to 60 cm interval, only two samples had detections of PCBs: Aroclors 1254 (11 µg/kg) and 1260 (14 µg/kg). These concentrations did not exceed the USEPA Region 4 soil screening value of 20 µg/kg. PCB concentrations were highest in Basin 3 below the depth where receptors would have contact with soils (i.e., greater than 60 cm).

8.1.5 Biological Tissue Characterization

Biota have been sampled in Onondaga Lake and its tributaries as part of the Onondaga Lake investigation by Honeywell/PTI/Exponent, as well as by NYSDEC for monitoring purposes. The plankton, benthic invertebrates, and fish data collected and analyzed are presented in the following section. Further information can be found in Chapter 5 of the RI (TAMS, 2002b).

8.1.5.1 Phytoplankton and Zooplankton

Zooplankton and phytoplankton were collected at two stations in Onondaga Lake in 1992 and were analyzed for methylmercury and total mercury. Total mercury and methylmercury were detected in all samples analyzed. These concentrations were converted to a wet-weight (ww) basis in the original data report (PTI, 1993b).

Methylmercury concentrations for phytoplankton, on a wet-weight basis, ranged from 4.3 to 39 µg/kg, and total mercury concentrations ranged from 85 to 300 µg/kg. Methylmercury concentrations for zooplankton, on a wet-weight basis, ranged from 21 to 184 µg/kg in combined zooplankton assemblages and 165 to 390 µg/kg in daphnids. Total mercury concentrations for zooplankton, on a wet-weight basis, ranged from 23 to 247 µg/kg in assemblages and 247 to 994 µg/kg in daphnids.

8.1.5.2 Benthic Macroinvertebrates

Total mercury and methylmercury were analyzed in benthic macroinvertebrates collected in Onondaga Lake in 1992 and 2000. Benthic organisms sampled in 1992 consisted of chironomids and amphipods. Benthic organisms sampled in 2000 consisted of chironomids, amphipods, and oligochaetes.

Total mercury was detected in all 1992 samples collected, with concentrations ranging from 268 to 2,500 µg/kg dry weight (dw). The maximum detected total mercury concentration in 1992 was found in a chironomid sample from Station S013, in the southeast corner of the lake between Onondaga Creek and Ley Creek, in a chironomid sample.

Methylmercury was detected in all 1992 samples collected, with concentrations ranging from 66 to 670 µg/kg dw (10 to 100 µg/kg ww). The maximum detected methylmercury concentration was found in an amphipod sample from Station S04 in the southwestern corner of the lake, near the mouth of Harbor Brook.

Total mercury was detected in all but one of the 2000 samples, with concentrations ranging from 187 to 53,200 µg/kg dw. Methylmercury was detected in 35 of 41 samples, with concentrations ranging from 17 to 2,500 µg/kg dw. The maximum concentrations for both mercury and methylmercury were detected at Station S406 in a chironomid sample, in the in-lake waste deposit between the East Flume and Harbor Brook. Total mercury concentrations were also elevated at Station S344 (35,500 µg/kg dw in an oligochaete sample) and at Station S404 (20,300 µg/kg dw in an oligochaete). These two stations are also in the vicinity of the East Flume and the Honeywell in-lake waste deposit.

8.1.5.3 Fish

Fish were collected from Onondaga Lake and its tributaries by Honeywell in 1992 and 2000. Data collected by NYSDEC between 1992 and 2000 are used in this BERA. All fish concentrations are given on a wet-weight basis.

Mercury and Other Metals

Methylmercury concentrations in fish receptors sampled by Honeywell and NYSDEC in Onondaga Lake and its tributaries in 1992 and 2000 ranged from 0.03 to 3.2 mg/kg ww. Mercury was detected in every fish analyzed. Summary of exposure concentrations by species can be found in Appendix H, Tables H-7 to H-14. The breakdown by species is as follows:

- Bluegill: 0.05 to 0.9 mg/kg.
- Gizzard shad: 0.07 to 0.4 mg/kg.
- Carp: 0.04 to 0.8 mg/kg.
- Channel catfish: 0.3 to 1 mg/kg.
- White perch: 0.2 to 2 mg/kg.
- Smallmouth bass: 0.3 to 1.7 mg/kg.
- Largemouth bass: 0.2 to 1.4 mg/kg (only mercury measured).
- Walleye: 0.3 to 3.2 mg/kg.

Since nearly all of the mercury in fish tissue consists of methylmercury, total mercury rather than methylmercury, was analyzed in the samples collected by Honeywell/Exponent in 2000.

Mercury, antimony, arsenic, chromium, selenium, vanadium, and zinc were the seven metals selected as COCs for fish receptors. Concentrations of metals other than mercury in fish are summarized below.

- **Antimony:** was detected in one catfish sample at a concentration of 1.8 mg/kg and in one white perch at a concentration of 2.1 mg/kg. It was undetected in the remaining species.

- **Arsenic:** was detected in bluegill, carp, and smallmouth bass. Concentrations ranged as follows:

  - Bluegill: 0.6 to 0.7 mg/kg.
  - Carp: 0.7 to 2 mg/kg.
  - Smallmouth bass: 1.1 to 1.8 mg/kg.

- **Chromium**: was detected in all receptor species in which it was analyzed. Concentrations ranged as follows:

  - Bluegill: 3 to 14 mg/kg.
  - Carp: 1.2 to 4.8 mg/kg.
  - Channel catfish: 1.3 mg/kg.
  - White perch: 0.6 mg/kg.
  - Smallmouth bass: 0.7 mg/kg.
  - Walleye: 0.7 mg/kg.

- **Selenium:** was detected in all receptor species in which it was analyzed, except the walleye. Concentrations ranged as follows:

  - Bluegill: 4.7 to 7.8 mg/kg.
  - Carp: 3.2 to 9 mg/kg.
  - Channel catfish: 4.8 to 5.7 mg/kg.
  - White perch: 3.5 mg/kg.
  - Smallmouth bass: 4.5 mg/kg.

- **Vanadium:** was detected in all receptor species in which it was analyzed, except for walleye and white perch. Concentrations ranged as follows:

  - Bluegill: 0.5 to 1.2 mg/kg.
  - Carp: 0.3 to 1 mg/kg.
  - Channel catfish: 0.8 to 1.1 mg/kg.
  - Smallmouth bass: 0.2 to 0.8 mg/kg.

- **Zinc:** was detected in all receptor species in which it was analyzed, except for walleye. Concentrations ranged as follows:

  - Bluegill: 35 to 108 mg/kg.

- Carp: 48 to 425 mg/kg.
- Channel catfish: 20 to 74 mg/kg.
- White perch: 17 mg/kg.
- Smallmouth bass: 35 to 56 mg/kg.

Organic COCs

DDT and metabolites, endrin, total PCBs, and dioxins/furans were selected as COCs in fish. Concentrations of these contaminants in receptor species are discussed below.

- **DDT and Metabolites**: was detected in all receptor species in which it was analyzed. Concentrations ranged as follows:

  - Bluegill: 11 to 28 μg/kg.
  - Carp: 15 to 300 μg/kg.
  - Channel catfish: 25 to 600 μg/kg.
  - White perch: 5 to 100 μg/kg.
  - Smallmouth bass: 2 to 240 μg/kg.
  - Largemouth bass: 2 to 84 μg/kg.
  - Walleye: 19 to 200 μg/kg.

- **Endrin:** was detected in all receptor species in which it was analyzed, except for largemouth bass. Concentrations ranged as follows:

  - Bluegill: 1.6 to 5.5 μg/kg.
  - Carp: 5.4 to 36 μg/kg.
  - Channel catfish: 6.2 to 46 μg/kg.
  - White perch: 12 μg/kg.
  - Smallmouth bass: 8.5 to 33 μg/kg.
  - Walleye: 6.5 μg/kg.

Total PCBs

While many fish were analyzed for PCBs by Honeywell in 1992, these data were not included in the BERA due to data quality issues (see BERA Chapter 11, Section 11.1.3). PCBs were detected in all receptor species analyzed by NYSDEC from 1992 to 2000 and by Honeywell in 2000. Concentrations ranged as follows:

- Bluegill: 300 to 875 μg/kg.
- Carp: 500 to 9,800 μg/kg.
- Channel catfish: 780 to 6,000 μg/kg.
- White perch: 370 to 3,800 μg/kg.

- Smallmouth bass: 210 to 11,000 μg/kg.
- Largemouth bass: 75 to 2,800 μg/kg.
- Walleye: 660 to 7,800 μg/kg.

Dioxins and Furans

Dioxin and furan data from fish collected by Honeywell/Exponent in 2000 and by NYSDEC in 1992, 1997, and 1999 were used in this BERA. TEQs were calculated based on risks to fish (van den Berg et al., 1998). Dioxins/furans were detected in all receptor species in which they were analyzed. Concentrations of dioxin/furan TEQs ranged as follows:

- Bluegill: 20 to 127 ng/kg-lipid.
- Carp: 34 to 1,055 ng/kg-lipid.
- Channel catfish: 38 to 286 ng/kg-lipid.
- White perch: 50 to 285 ng/kg-lipid.
- Smallmouth bass: 26 to 165 ng/kg-lipid.
- Largemouth bass: 146 to 393 ng/kg-lipid.

8.2 Exposure Assessment

The assumptions and models used to predict the potential exposure of plants, fish, and wildlife (i.e., mammals and birds) to COCs associated with Onondaga Lake are described in this section. Site-specific chemical data characterizing the distribution of COCs in prey items (i.e., fish) and modeling the distribution in other prey items (aquatic invertebrates, terrestrial invertebrates, and small mammals) are discussed here. Food-web models used to estimate exposure of wildlife receptors to COCs are described, along with receptor life history characteristics and exposure assumptions. Receptors discussed in this section are surrogates for all species that inhabit or may inhabit Onondaga Lake.

8.2.1 Definition of Assessment Units

For the risk analyses, three specific assessment units were considered to represent the various receptor habitats. Sampling data from Onondaga Lake were divided into subsets depending on the location and habitat characteristics in order to evaluate receptor exposure. The three assessment units were defined as follows:

Onondaga Lake – The lake was divided into two areas, the pelagic and littoral areas. The pelagic assessment area encompasses the lake water column from the surface to the thermocline. All biota inhabiting this region were considered part of the pelagic assessment area. The littoral assessment area was considered to be the nearshore habitat from the edge of the lake to the point where the water depth exceeded 2 m. Sediments within the assessment area were considered to be from the surface to 15 cm for the evaluation of both incidental sediment ingestion and modeling uptake by biotic prey items. All biota within both the water column and the sediments were included in the Onondaga Lake assessment unit.

Wetlands – The wetland assessment unit focused on wetlands with hydrological connections to the lake. Data were collected from four of the wetlands that exist along the lake, including wetland areas along the northwest shoreline of the lake (Wetland SYW-6), and at the mouths of Ninemile Creek (Wetland SYW-10), Harbor Brook (Wetland SYW-19), and Ley Creek (Wetland SYW-12). All wetland areas were evaluated separately due to the locations and characteristics of each area, which suggest that contaminant concentrations and associated risks are likely to differ. The two northern basin wetlands (SYW-6 and SYW-10), both of which are on the west side of the lake between Ninemile Creek and the lake outlet, are dominated by floodplain forest, emergent swamps, and emergent vegetation (see Chapter 3, Section 3.2.4.1). These wetlands are expected to more closely resemble each other due to the general similarity of the area in which they are located. Wetland SYW-10 is being further investigated by Honeywell and NYSDEC as part of the Geddes Brook/Ninemile Creek RI/FS.

The southern Wetland SYW-12 is dominated by emergent vegetation as it approaches the lake, and along the shore of the lake it is a combination of floodplain forest and emergent marsh. Wetland SYW-19 is dominated by reedgrass and is severely contaminated with Honeywell COCs (note that this wetland is also being investigated as part of Honeywell's Wastebed B/Harbor Brook site).

Wetland receptors include plants, carnivorous birds, and insectivorous mammals. Avian and mammalian receptors were considered to have incidental soil exposure via ingestion up to a depth of 15 cm, based on the depth of the surface soils sampled in 2000. Likewise, soils from 0 to 15 cm were used for modeling the concentrations of contaminants in mammalian prey, as small mammals were considered to have the greatest exposure to this depth profile.

Dredge Spoils Area – The upland region represented in this BERA includes areas adjacent to the northwest shore where dredge spoils were used as reclamation fill in the late 1960s. Receptors in this unit include plants, carnivorous birds, and insectivorous mammals. Avian and mammalian receptors were considered to have incidental soil exposure via ingestion down to a depth of 107 cm. Likewise, all surface soils collected, ranging from 0 to 107 cm, were used for modeling the concentrations in mammalian prey. Drinking water was assumed to come from Onondaga Lake.

All receptors modeled for this assessment were determined to have foraging ranges within the Onondaga Lake area and were considered to be closed populations. Therefore, data from surrounding sites were not used directly in this BERA. However, other sites surrounding Onondaga Lake may represent additional sources of potential exposure. These other sites are discussed in Chapter 2 and Appendix G of this BERA.

8.2.2 Calculation of Exposure Point Concentrations

This section presents the methodology that was employed to calculate exposure point concentrations (EPCs) for the COCs in each exposure area. For each data set (representing a single chemical in one medium in an exposure area), a 95 percent upper confidence limit (UCL) on the mean concentration was calculated and compared to the maximum detected concentration for that chemical. The lower of the 95 percent UCL and the maximum detected value was used as the exposure point concentration, as recommended by USEPA (USEPA, 1992b).

Given a data set with no non-detect values, calculation of the UCL is straightforward. Depending on whether the data are normally or lognormally distributed, the UCL can be calculated using the Student's t-statistic or the H-statistic, respectively (USEPA, 1992b). In the presence of non-detects, however, the calculation is more complicated. The mean and standard deviation of the data (or the log-transformed data for lognormally distributed data sets) must be estimated in order to calculate the UCL.

Step 1: Assign Values to Non-Detect Data Points

USEPA guidance (1989) specifies that one-half the detection limit be used for non-detected results (i.e., data qualified "U") by the laboratory. Although the procedure is straightforward for inorganics (metals and cyanide), the determination of the appropriate value to use as the detection limit for organics (volatiles and semivolatiles) is open to question. However, based on USEPA Region 2 direction, all EPCs used in this assessment were based on non-detected results for organic and inorganic compounds being assigned a value of half the laboratory-reported detection limit (i.e., one-half the "U" value).

Step 2: Determine Data Distribution Type (Normal or Lognormal)

In accordance with USEPA (1992b]), the type of data distribution exhibited by a compound of concern in a medium (specifically, normal or lognormal) was evaluated based on a calculation of the W-test statistic developed by Shapiro and Wilks (1965) for sample sets containing more than 10 and less than 50 samples. The W-test was applied to each COC in each medium. This test is designed to examine the likelihood that the underlying population is normally distributed based on a random sample set. See Gilbert (1987) for details of the calculation method.

Values for W lie between 0 and 1. The closer the W value is to 1.0, the more normally distributed the data set is. The W-statistic was calculated for the data using the non-detect substitutions described in Step 1 and the log-transform of these distributions. If the W for the log-transformed data was greater than the W of the untransformed data, the distribution was assumed to be lognormal. Conversely, if the W for the untransformed data was greater than the W of the log-transformed data, the distribution was assumed to be normal. Where the W-statistic for the transformed and untransformed data was identical (such as when only two samples were collected), the distribution was assumed to be lognormal for the purpose of calculating the UCL.

Step 3: Calculate Exposure Point Concentrations

The EPCs used in this risk assessment were the arithmetic mean and an upper-bound estimate based on the lower value of the maximum detected value and the 95 percent UCL on the mean (see Appendix H). The term "95 percent UCL" is used throughout the remainder of the BERA to represent the upper-bound estimate. The UCL was calculated from the summary statistics, depending on the form of the distribution that best fits the data.

For normally distributed data sets, the UCL on the mean is calculated as:

$$UCL = \overline{X} + t\left(\frac{s}{\sqrt{n}}\right)$$

where:

$\overline{X}$	=	arithmetic mean of the sample data set for the compound of concern.
s	=	sample standard deviation of the sample data set for the compound of concern.
t	=	the Student's t-statistic for the 95 percent confidence interval for a one-tailed distribution; the t-statistic is a function of the number of samples collected.
n	=	number of samples in the data set.

For lognormally distributed data sets, the UCL on the mean is calculated as:

$$UCL = \exp\left[\overline{X} + 0.50s^2 + \frac{Hs}{\sqrt{n-1}}\right]$$

where:

$\overline{X}$	=	arithmetic average of the natural log-transformed data.
s^2	=	variance of the log-transformed data.
s	=	sample standard deviation of the log-transformed data.
H	=	H-statistic; the H value differs from the t-value because the formula is designed to estimate the UCL on the basis of the log-transformed data. H is a function of the standard deviation of the log-transformed data and the number of samples in the data set. H was taken from a standard table of calculated values (Gilbert,

1987) or linearly interpolated between values given in the table where necessary.

n = the number of samples in the data set.

When the data set contains less than ten sample results, the EPC was the maximum detected concentration.

Field Duplicates

Field duplicates were averaged based on USEPA Region 2 protocols as follows:

When averaging a set of data:

- Do not use rejected values. Do not include them in the sample count used to calculate the average.
- If the parameter was not detected, use half the detection limit.

When averaging data from a duplicate and its original sample:

- Detect versus detect: Average the two values and combine the qualifiers.
- Detect versus non-detect: Use only the detect. Note that the other value was not detected. Do not use half the detection limit.
- Detect versus rejected value: Use the detect. Discard the rejected value. Note the rejected value was not used.
- Non-detect versus non-detect: Average the non-detect values using half the detection limit.

Chemical of Concern Groups

Non-detected values were treated as observations at one-half the detection limit, with the exception of contaminants representing a group of COCs, such as PCBs and PAHs. Concentrations of PCBs were calculated as follows:

- When two or more Aroclors were detected, total PCBs was calculated to be the sum of all detected Aroclors.
- If only one Aroclor was detected, total PCBs was calculated to be the sum of the detected Aroclor concentration, plus half the detection of one other Aroclor (note:

all Aroclors [except 1221, which was not detected] have the same detection limit, so it is irrelevant which Aroclor is used).

- If no Aroclors were detected, total PCBs was calculated to be the sum of half the detection limit of each of two Aroclors.

Concentrations of PAHs were calculated as the sum of all detected PAHs. The sum of all detected compounds in a group was also used to calculate EPCs for DDT and metabolites, dichlorobenzenes, trichlorobenzenes, chlordanes, heptachlor/heptachlor epoxide, endosulfans, hexachlorocyclohexanes, and dioxins/furans.

8.2.3 Exposure Characterization for Aquatic Plants and Invertebrates

Risks to aquatic macrophytes and phytoplankton could not be assessed by comparing the mean and 95 UCL concentrations of COCs measured in surface water to water quality standards, criteria, and guidance, as there are no standards that specifically address the risk to for aquatic plants. However, narrative water quality standards (6 NYCRR Part 703.2), which regulate physical parameters and aesthetic conditions that impair the best use of the surface water but may not be physically measurable, were used to qualitatively evaluate water quality effects on aquatic macrophytes and phytoplankton. The effects of SOCs on aquatic plants were evaluated using site-specific literature.

Risks to zooplankton and benthic invertebrates were assessed by comparing the mean and 95 UCL concentrations of COCs measured in surface water and sediments (as appropriate based on the receptor) to water and sediment quality criteria for aquatic organisms. Sediment toxicity to benthic invertebrates was also evaluated using the results of laboratory toxicity tests conducted with Onondaga Lake sediments, as discussed in Chapter 9. The effects of SOCs were evaluated using site-specific literature. Aquatic invertebrates were considered to be part of the Onondaga Lake assessment unit.

8.2.4 Exposure Characterization for Terrestrial Plants

Plants were evaluated separately for the wetlands and the dredge spoils assessment units, since the two areas have different plant communities (see Appendix A and Chapter 3, Figure 3-4). The wetland areas are vegetated with floodplain forest, emergent vegetation, or reedgrass, or a combination of these covertypes. Wetland samples were collected from four wetlands connected to the lake: SYW-6, SYW-10, SYW-12, and SYW-19 (see Chapter 3, Section 3.2.4.1). Trees and other plants have colonized the dredge spoils area.

The same COCs were evaluated for both habitats, although concentrations of contaminants varied somewhat between locations. All of the COCs identified in the screening assessment as potentially posing a risk to plants are natural constituents of soils (i.e., inorganics), but this is partially due to the lack of plant screening values for organic compounds. Soil contaminated with heavy metals can produce apparently normal plants that may be unsafe for human or animal consumption (Kabata-Pendias and Pendias, 1992).

The primary potential for adverse impact to plants or plant communities from a COC is related to its uptake availability through the roots. For uptake to occur, a COC must be water-soluble and capable of being transported symplastically across the Casparian strip. The screening concentrations used are nominal concentrations of a soluble form (i.e., a highly bioavailable form) of the chemical added to soil (Efroymson et al., 1987a). Most metals in natural soils and contaminants of waste sites are in not readily bioavailable forms; therefore, risk estimates are considered conservative, as discussed in Chapter 11, Uncertainty Analysis.

8.2.5 Exposure Characterization for Fish

One method used to assess potential threats to fish in this BERA was to compare concentrations of COCs identified in the screening against TRVs selected from the literature. Site-specific literature was evaluated to determine the effects of SOCs. Fish were considered to be part of the Onondaga Lake assessment unit.

Forage fish, planktivorous fish, omnivorous fish, and piscivorous fish from Onondaga Lake were analyzed for contaminants. An overview of the biology, habitat selection, and feeding habits of fish receptors sampled in the lake are discussed in this section. Fish species selected as receptors represent a variety of habitats, sources of food, longevity, and size, all of which are likely to contribute to the amount of contamination that they come into contact with and subsequently accumulate.

Fish may be exposed to contaminants via direct uptake from the water column, uptake from sediments, and through feeding. Fish that feed extensively on fish eggs (e.g., white perch) contribute to a closed contaminant transfer loop between fish eggs and fish in Onondaga Lake.

Species profiles of fish receptors discussed in this BERA are presented below. The information presented is taken primarily from Werner (1980), Smith (1985), and Scott and Crossman (1973) or is based on professional judgment.

8.2.5.1 Bluegill (*Lepomis macrochirus*)

Bluegill are found in warm shallow waters in ponds, lakes, and in slow-moving bodies of water where there is adequate vegetation or other shelter in summer. In the winter they may retreat to deeper, cooler water where they tend to remain in colonial groups. Spawning occurs in late spring to mid-summer in colonial-style nests that are sometimes relatively dense. Growth is rapid, and bluegill can grow up to 25 to 28 cm in length, with ages up to 11 years.

Bluegill feed throughout the water column during the day, primarily in the morning and afternoon. They are omnivorous and eat a wide variety of organisms and, at times, significant amounts of plant material. Young bluegills feed on rotifer and copepod nauplii, while larger individuals eat insects and other larger particles.

8.2.5.2 Gizzard Shad (*Dorosoma cepedianum*)

Gizzard shad is predominantly a quiet-water fish, although it has been collected it in swift streams in the Genessee drainage in New York. It can tolerate high turbidity and relatively high salinities of 33 to 34 ppt, but most often is found in clear water. It is often found near the surface, and its young are common around weed beds. The young school during their first year and are sometimes found well upstream in small streams.

Newly hatched gizzard shad feed on protozoans and other zooplankton. After a few weeks the diet changes to include phytoplankton and algae. The gizzard shad is essentially a filter feeder. Lake Erie populations of gizzard shad may reach six years of age (five is more common) and lengths slightly greater than 38 cm (Scott and Crossman, 1973).

8.2.5.3 Carp *(Cyprinus carpio)*

Carp inhabit lakes, ponds, and larger streams and are most abundant where there is dense aquatic vegetation. Carp spawn in the spring and early summer when temperatures reach about 17°C (63°F) (Scott and Crossman, 1973). A female will lay from 36,000 to over two million eggs of approximately 1 mm in diameter. These adhesive eggs are deposited randomly in shallow waters, generally over aquatic vegetation. Growth is rapid after hatching, which occurs in about three to six days.

Carp are omnivorous bottom feeders and feed on filamentous algae and various benthic invertebrates, including snails, annelids, midge larvae, and crustaceans. In their feeding activities they often destroy vegetation by physically uprooting the plants, stirring up the bottom, and by making the water so turbid that sufficient light cannot reach the growing plants. Occasionally they may move up in the water column to feed on plant and animal materials. Carp may reach up to 18 cm in their first growing season and have been observed up to 86 kg in Lake Erie (Scott and Crossman, 1973). They are long-lived fish and 20-year-old fish are considered normal in North America.

8.2.5.4 Channel Catfish (*Ictalurus punctatus*)

The channel catfish occurs in larger streams, rivers, and lakes, where it is able to thrive in moderate currents over sandy to rocky substrate. It can tolerate relatively low oxygen levels and warm water, and has been found in waters with oxygen as low as 0.95 ppm and temperatures in excess of 32°C (90°F) (Smith, 1985). Channel catfish are not normally associated with aquatic vegetation as are other members of the catfish family that inhabit Onondaga Lake, such as brown bullhead (*Ameiurus nebulosus*) and yellow bullhead (*Ameiurus natalis*).

The channel catfish is a nocturnal feeder and depends heavily upon chemical senses to locate its food. The young feed largely on aquatic insects and other bottom-dwelling arthropods. When they reach about 100 mm standard length they become omnivorous, with fish making up a large part of their diet (Smith, 1985). Seeds and terrestrial animals, including birds, have been found in catfish stomachs. In southern ranges channel catfish may reach ages over 20 years and sizes over 50 cm; however, in western Lake Erie, ages only extend to seven years, and larger fish are around 36 cm (Scott and Crossman, 1973).

8.2.5.5 White Perch (*Morone americana*)

White perch can tolerate a wide range of salinities from marine to fresh water. White perch have been known to migrate to shallows or to surface waters at night and offshore or to deeper waters during the day. They congregate in areas with DO of at least 6 mg/L (Seltzer-Hamilton, 1991) and are often found in rather turbid shallow areas where at times they form dense schools.

Young white perch feed heavily on small invertebrates, such as copepods, during their first two summers. *Gammarus*, chironomid larvae, and occasional *Cyathura* also become important foods as they grow (Smith, 1985). Fish eggs are an important food source in late spring and early summer (May through July). White perch more than 200 mm in length feed mostly on other fish (Setzler-Hamilton, 1991). Spawning takes place in the spring in waters between 14 and 16°C (58 and 60°F). Females have relatively large numbers (20,000 to 300,000) of small eggs (0.55 to 0.70 mm). White perch have an average life span of five to seven years and have been reported as old as 17 years. Growth rates vary widely and sizes over 33 cm and 3.8 kg have been documented (Scott and Crossman, 1973).

8.2.5.6 Smallmouth Bass (*Micropterus dolomieui*)

Smallmouth bass are found in streams and lakes. Although they tolerate a wide range of habitats, they tend to select cool waters with rocky or gravel substrate where there is shelter. Smallmouth bass spawn in May or early June. A mature female smallmouth bass may lay a total of 5,000 to 7,000 eggs in several nests that the males build and guard (Werner, 1980).

Juvenile smallmouth bass feed on plankton and invertebrates, switching to larger items as they grow. Smallmouth bass are opportunistic predators and feed on primarily on insects, crayfish, and fish, but will also feed on amphibians (e.g., tadpoles, frogs, and salamanders) and small animals, if available. In Lake Erie, smallmouth bass live up to nine years and reach sizes up to 38 cm (Scott and Crossman, 1973).

8.2.5.7 Largemouth Bass *(Micropterus salmoides)*

The largemouth bass is a relatively large, robust fish that has a tolerance for high temperatures and slight turbidity (Scott and Crossman, 1973). Largemouth bass occupy warm, weedy parts of lakes, ponds, and streams and show a low tolerance for low oxygen conditions. Largemouth bass mature at age five and spawn from late spring to mid-summer, in some cases as late as August. Male largemouth bass construct nests in sand and/or gravel substrates in areas of non-flowing clear water containing aquatic vegetation (Nack and Cook, 1986). Females produce 2,000 to 7,000 eggs per pound of body weight (Smith, 1985) and leave the nest after spawning.

Until they are about 5 cm in length, young-of-year (YOY) feed on plankton, insects and other invertebrates. As they get larger, their diet shifts to fish and other large items including almost anything that moves, including amphibians, reptiles and terrestrial species. Largemouth bass longer than 50 mm total length usually forage exclusively on fish. The largemouth bass represents a top predator in the aquatic food web, consuming primarily fish, such as gizzard shad, carp, bluntnose minnow (*Pimephales notatus*), golden shiner (*Notemigonus crysoleucas*), yellow perch (*Perca flavens*), pumpkinseed (*Lepomis gibbosus*), bluegill, and other largemouth bass (Scott and Crossman, 1973).

Largemouth bass take their food at the surface during morning and evening, in the water column during the day, and from the bottom at night. They feed by sight, often in schools, nearshore, and almost always close to vegetation. Feeding is restricted at water temperatures below 10°C (50°F) and decreases in winter and during spawning. Largemouth bass do not feed during spawning. In Lake Ontario largemouth bass live up to 13 years and reach sizes that average about 50 cm.

8.2.5.8 Walleye (*Stizostedion vitreum*)

Walleye occur in lakes and larger rivers and are active year-round. They generally swim near the bottom in loose aggregations and frequently move into shallows to feed at night. Spawning occurs in early spring over coarse gravel shoals in lakes, or over gravel and rocky bottoms in rivers or tributaries to the lakes the walleye inhabit. Eggs are 1.5 to 2 mm, with numbers from large females often as high as high as 500,000 (Scott and Crossman, 1973). An estimated 20 percent of the eggs survive to hatching under ideal conditions, and less than 5 percent is common under suboptimal conditions. To achieve a stable population only a small percent of those eggs surviving need to mature and reproduce (Werner, 1980).

Walleye are opportunistic predators. Upon hatching, walleye feed first on plankton crustaceans, but soon switch to insects and then to fry, including other walleye if food is scarce or there is crowding. By the time they are 8 cm long, walleye feed on fish and other larger items. Walleye live from 10 to 12 years in southern Canadian waters (including Lakes Erie and Ontario) and reach trophy sizes of between 12 to 19 pounds in Lake Ontario.

8.2.6 Exposure Characterization for Terrestrial Wildlife

Exposure of terrestrial wildlife to the COCs in Onondaga Lake was determined using a food-web modeling approach. Site-specific daily doses were estimated for each of the receptors based on their expected COC exposures resulting from modeled rates of contact with specific media. This approach allows for a direct comparison of exposure to toxicity in the characterization of the risk posed by the COC to receptor populations. Wildlife populations are defined as all individuals of a receptor species who may be exposed to COCs associated with Onondaga Lake water, sediment, soil, or biota.

Total exposure for receptor populations was determined through the summation of all pathways of exposure. It was assumed that the exposed receptor population is completely closed (i.e., no interactions with any population or location outside of the lake itself), and as such, dietary, drinking water, and incidental sediment ingestion is derived from the appropriate assessment area of Onondaga Lake for their entire lifetime. Exposure to contaminated areas outside the Onondaga Lake area is discussed in the uncertainty section.

8.2.6.1 Food-Web Modeling

A deterministic risk assessment was performed to characterize risk to receptors from exposure to COCs. The exposure rate was predicted based on the mean and 95 percent UCL on the mean (or the maximum if less than the UCL) for COC concentrations measured in Onondaga Lake and was interpreted to be representative of exposed populations.

The general structure of the model used to estimate the exposure rate for a given contaminant by a wildlife receptor is as follows:

$$\mathrm{EED} = \sum \left(\mathrm{IR}_p \times [\mathrm{COC}]_p + \mathrm{IR}_w \times [\mathrm{COC}]_w + \mathrm{IR}_s \times [\mathrm{COC}]_s \right)$$

where:

EED	=	estimated environmental dose (mg/kg body weight-day)
IR_p	=	receptor-specific prey intake rate (kg dry weight/kg body weight)
IR_w	=	receptor-specific water intake rate (L/kg body weight)
IR_s	=	receptor-specific incidental sediment intake rate (kg dry weight/kg body weight)
$[\mathrm{COC}]_p$	=	COC concentrations in the receptors' prey (mg/kg dry weight)
$[\mathrm{COC}]_w$	=	COC concentrations in the receptors' drinking water (mg/L)
$[\mathrm{COC}]_s$	=	COC concentrations in the sediments or soils incidentally ingested (mg/kg dry weight)

Derivations of the parameters used to predict the exposure doses are discussed in the following sections.

8.2.6.2 Routes and Media of Exposure

The route of exposure is defined as the means by which a receptor may contact a contaminated medium. For this assessment, exposure was limited to ingestion because it was assumed to account for the majority of exposure to the COCs.

Based on the chemical properties of the COCs and the typical foraging behavior of the receptors, it was concluded that the primary routes of exposure of wildlife to COCs would be through: 1) ingestion of prey items (e.g., macroinvertebrates/insects, fish, small mammals), 2) ingestion of drinking water, and 3) the incidental ingestion of soil or sediment.

8.2.6.3 Wildlife Receptor Assessment Unit Association

Wildlife receptors were selected to represent species that inhabit or may inhabit Onondaga Lake. Birds selected were the tree swallow (*Tachycineta bicolor*), mallard duck (*Anas platyrhynchos*), belted kingfisher (*Ceryle alcyon*), great blue heron (*Ardea herodias*),osprey (*Pandion haliaetus*), and red-tailed hawk (*Buteo jamaicensis*). Mammals selected as receptors are the little brown bat (*Myotis lucifugus*), short-tailed shrew (*Blarina brevicauda*), mink (*Mustela vison*), and river otter (*Lutra canadensis*).

These receptors do not cover the entire range of species found around Onondaga Lake (see Chapter 3), but were selected to represent the species potentially at risk based on their exposure to specific prey items (e.g., piscivorous, insectivorous) and habitats associated with the lake. Receptors feeding on items with lower contaminant concentrations, such as herbivores (e.g., muskrat, deer mice), are at lower risk than receptors feeding on higher trophic level prey, and, therefore, this risk assessment is considered to be protective of them, as discussed in the preliminary conceptual model (Chapter 4, Section 4.1). The receptors associated with each of the assessment units (Section 8.2.1) are as follows:

- **Onondaga Lake – Pelagic Habitat**: The receptors expected to be at greatest risk in this habitat are those that forage within the water column of the open lake. There are no mammalian species indigenous to this region that utilize this habitat. However, the osprey does hunt in the pelagic zone, and therefore may be exposed to COCs in this region of Onondaga Lake. An intermediate case of exposure to the pelagic region was considered for the tree swallow and little brown bat, which feed predominantly on emergent insects. Prey items for these receptors were assumed to originate from anywhere within the entire lake. Other receptors in this unit include macrophytes, phytoplankton, zooplankton, and fish.

- **Littoral Habitat**: The receptors expected to be at greatest risk in this habitat are those that forage within the inshore zone of the lake and are dependent upon indigenous aquatic organisms as their primary food source. The terrestrial receptors considered most likely to be at risk are those that use the lake as a prey source, but are not expected to venture beyond the immediate shoreline in search of prey. The receptor species considered for this assessment unit are the mink, river otter, belted kingfisher, great blue heron, and mallard. Other receptors in this unit include macrophytes, benthic invertebrates, fish, and insectivorous birds and mammals.

- **Wetlands:** The receptors expected to be at greatest risk in this habitat are those that forage on insects and small mammals. These receptors are the short-tailed shrew (native insectivore) and red-tailed hawk (native top carnivore).

- **Dredge Spoils Area:** As in the wetlands habitat, the receptors expected to be at greatest risk in this habitat are those that forage on insects and small mammals. These receptors are the short-tailed shrew and red-tailed hawk.

8.2.6.4 Chemical of Concern Exposure from the Ingestion of Fish

Prey selection is a function of the receptor's size and method of hunting. Prey selection plays an important role in modeling exposure, because some prey sizes (e.g., large fish) may have higher concentrations of contaminants than others. For assessment purposes, prey selection was refined to provide greater confidence in the data sets gathered (i.e, increased sample size). The two parameters that often account for the much of the variation seen in contaminant concentrations in fish from a single location are species (feeding patterns, habitat) and age of the fish (longer period of bioaccumulation, change in feeding patterns). In general, the higher an organism is on the food chain and the older it is, the greater the concentration of bioaccumulative contaminants.

All the fish within the receptor prey size-selection range, for which COC concentration estimates were available, were used to predict the receptor's exposures. Fish consumed by wildlife receptors were divided into two size classes (3 to 18 cm and 18 to 60 cm) based on the available data and prey selection preferences of receptors.

Whole Fish and Fillet Data

Avian and mammalian receptors were assumed to consume whole fish. Therefore, ratios were developed to convert fillet concentrations to whole fish concentrations. Estimates of whole body COC concentrations were expressed on a dry weight basis in all exposure models to control for variations in water content between fish and between fish and sediment. The standardization also permitted direct application to ingestion rates that were determined in dry weight.

Small fish, such as bluegills, were generally analyzed as whole fish samples and contaminant body burdens could be used directly for ecological modeling. Omnivorous fish were analyzed as both whole fish and fillets. The majority of data available for piscivorous species is based on analyses of fish fillets.

Fish tissue concentrations used in this BERA were collected in 1992 and 2000 by Honeywell and between 1992 and 2000 by NYSDEC. Data from both Honeywell and NYSDEC were pooled into a single data set, which was then queried for information for evaluation of fish as receptors and size classes for use in food-web modeling (i.e., fish as prey of piscivorous receptors). NYSDEC has analyzed mainly fillets from species caught by anglers such as smallmouth bass, largemouth bass, white perch, and walleye, while

Honeywell's analyses were taken from a mix of species with various feeding habits. Therefore, Honeywell data were used to derive fillet to whole fish ratios.

Honeywell analyzed the fillet and remains from 11 fish (seven from Onondaga Lake and four from lower Ninemile Creek) in 2000 and used the data to develop regressions for estimating whole-fish concentrations from fillet concentrations. The 11 samples for which the fillet and remains were available consisted of two bluegill, two catfish, five carp, and two smallmouth bass. Although carp comprised about 45 percent of the fillet-remains data, it was only about 12 percent of the total Honeywell mercury samples and about 1 percent of the NYSDEC mercury samples. The regressions developed by Honeywell were considered inappropriate for use in this BERA based on the small number of samples used for the regression, the low correlation coefficients of some of the regression equations, and variability introduced by the use of several species with different lipid concentrations, feeding patterns/trophic levels, habitats, and other variables.

NYSDEC/TAMS developed conversion factors for contaminants with fillet and whole fish data for use in this BERA (Table 8-4). Whole-body concentrations were determined from the separate analyses of fillet and remains using the following formula:

$$[COC]_{\text{Whole Body}} = \frac{[COC]_{\text{fillet}} \times \text{Mass}_{\text{fillet}} + [COC]_{\text{Remain}} \times \text{Mass}_{\text{Remain}}}{\text{Mass}_{\text{fillet}} + \text{Mass}_{\text{Remain}}}$$

Fillet to whole fish conversion factors were used for mercury, total PCBs, DDT and metabolites, and dioxins/furans. A default value of one was used for other organic COCs, due to small sample sizes or low detection rates (e.g., hexachlorocyclohexane), and for all metals exclusive of mercury (e.g., arsenic, chromium, selenium, vanadium, and zinc), which had high levels of uncertainty associated with calculated ratios.

- **Mercury** – Honeywell fillet and whole-fish data (n = 22 and 11, respectively) yielded a ratio of 1.1. USEPA performed a national survey of mercury in fish in which a fillet to whole fish conversion factor of 0.7 was calculated (USEPA, 1999d). Scientists at ORNL also calculated a conversion factor of 0.7 for mercury (Bevelhimer et al., 1997). Based on the consensus for mercury found in the literature, a conversion factor of 0.7 was used to calculate the concentration of mercury in whole fish based on fillet samples (i.e., mercury in whole fish = mercury in fillet × 0.7).

- **PCBs** – PCBs tend to bioaccumulate in fatty (lipid-rich) tissues, resulting in higher PCB concentrations in whole fish than fillets, in contrast to mercury where retention of mercury in muscles and other tissues resulted in higher fillet concentrations. PCB concentrations were higher in whole fish than fillets. Although the Honeywell PCB data set was much smaller than the NYSDEC data set (11 fillets versus 112 fillets), only the Honeywell data set was used to calculate ratios in order to compare similar species, as different species vary in lipid content and other parameters that influence total PCB concentration. Seventeen whole fish were used to calculate the ratio.

A wet-weight comparison resulted in a conversion ratio of 2.5 from fillet to whole-fish concentrations. This number was compared to the conversion used in the Hudson River PCBs site Baseline Ecological Risk Assessment (TAMS/USEPA, 2000) to confirm if it was representative. A ratio of 2.5 was obtained for the largemouth bass and a ratio of 1.5 was obtained for the brown bullhead in that assessment. The Onondaga Lake ratio was determined to correspond to other freshwater systems, and a ratio of 2.5 was applied to convert fillet concentrations to whole-fish concentrations.

- **DDT and Metabolites** – A ratio of 2.3 was calculated for DDT and metabolites based on 11 fillets and 17 whole fish collected by Honeywell. Species used for analyses were smallmouth bass, bluegill, carp, and catfish. The Honeywell regression equation used in the 2000 BERA (TAMS/USEPA) is not considered to be appropriate because concentrations of DDT in fish were generally less than 0.1 mg/kg and the regression equation overestimates DDT concentrations in that range.

- **Dioxins and Furans** – Ratios of 1.7 and 1.8 were calculated for dioxins and furans on a TEQ basis for avian and mammalian receptors, respectively. Eleven fillets and 18 whole fish samples were used to derive the ratios.

8.2.6.5 Chemical of Concern Exposure from the Ingestion of Terrestrial Prey

In estimating exposure rates for receptors consuming terrestrial prey items (i.e., mink and red-tailed hawk), COC concentrations were modeled based on available measured concentrations in wetland soils/sediments and surface soil concentrations (dry-weight basis). All non-detected values were considered observations at one-half the detection limit.

COC concentration in the prey items was predicted through the application of COC-specific transfer factors derived from ORNL guidance documents (Sample et al., 1998a,b) provided in Table 8-5. These factors are based on concomitant analyses of COC concentrations in both soil and appropriate biological tissues. COC concentrations ($[COC]_{prey}$) were modeled based on available estimates of soil concentrations ($[COC]_{soil}$) on a dry-weight basis. This modeling was accomplished through application of a COC-specific transfer factor ($TF_{soil \rightarrow prey}$) as follows:

$$[COC]_{prey} = [COC]_{soil} \times TF_{soil \rightarrow prey}$$

Prey were grouped into two classes based on the feeding patterns of receptors. The first class was soil invertebrates, which were represented in this assessment by earthworms and serve as prey for receptors, which were represented by the short-tailed shrew. The second class was small terrestrial mammals, which were defined for assessment purposes as any herbivorous, omnivorous, or insectivorous species (less than 2 kg in mass) that may be potential prey for receptors, which were represented by the red-tailed hawk and mink.

Concentration-dependent regressions or representative transfer factors were applied to calculate mean and 95 percent UCL contaminant concentrations in prey. Non-detected values were considered to be observations at one-half the detection limit. In contrast to the screening assessment, general regressions or median UFs were selected for the baseline assessment rather than the 95 percent upper prediction limit (UPL) or 90th percentile UF for both earthworm and small mammal models. The only exception to this procedure was methylmercury, for which no UFs were available, and therefore the conservative recommendation for mercury was applied.

Data from muskrats trapped in the vicinity of Geddes Brook (GB) and Ninemile Creek (NMC) between July 1998 and November 1998 were not used to develop transfer factors for mercury, PCBs, and dioxins and furans in small mammals. Prior to collection, NYSDEC eliminated the muskrat sampling effort from the GB/NMC field investigation (NYSDEC/NYSDOL, 1998) on the basis that it was inappropriate to use a herbivorous mammal to represent small mammals, inclusive of insectivores, based on differences in bioaccumulation related to feeding strategies.

In addition, the reliability of the transfer factors is questionable. The soil-muskrat transfer factors developed by Honeywell were based on seven muskrats from three locations, and six of the muskrats were collected from reference stations. The transfer factors have significant variability associated with them due to the small sample size and the narrow range of contaminant concentrations. In particular, the transfer factor for PCBs is considered to be particularly unreliable since levels of PCBs were below detection limits in muskrats and there was only one detection of PCBs (Aroclor 1254) in one of the co-located soil samples, resulting in the PCB uptake factor being based primarily on half the detection limit. Hence, transfer factors from the literature were considered to be more reliable than the Honeywell factors and were used in this assessment.

8.2.6.6 Chemical of Concern Exposure from the Ingestion of Benthos or Emergent Insects

COC exposure concentrations for receptors feeding upon emergent insects (i.e., tree swallow and little brown bat) and benthic macroinvertebrates (mallard duck) were modeled using a transfer factor method. The exception was mercury (methylmercury and ionic mercury) for which adequate measured invertebrate observations from the lake were available. Three amphipod samples were analyzed for PCBs in 1992, which was not sufficient to estimate PCB concentrations in macroinvertebrates. In addition, there were quantitation uncertainties associated with the Honeywell 1992 PCB analyses in biological tissue, as discussed in the uncertainty analysis (Chapter 11, Section 11.1.3).

In the transfer factor models, estimates of COC concentration in benthic larvae ($[COC]_{insects}$) were derived from measured concentrations in sediments ($[COC]_{sediment}$ on a dry-weight basis). Predicted COC concentrations were determined using available biota-sediment accumulation factor (BSAF) values (TF_{BSAF}) as predictive transfer coefficients, as follows:

$$[COC]_{insect} = [COC]_{sediment} \times TF_{BSAF}$$

BSAFs for metals were taken from recommendations from the Oak Ridge Reservation (US Department of Energy [USDOE], 1998). In contrast to the screening assessment, the degree of overestimation was minimized by using general, rather than conservative, recommendations.

All BSAFs for organics are from the US Army Corps of Engineers (USACE) BSAF database (USACE, 2002) or based on professional judgment. Freshwater invertebrate data were used from the USACE database, when available. The BSAFs from all freshwater invertebrate serving as prey for receptors in this assessment averaged for each contaminant to obtain contaminant-specific BSAFs. If no freshwater invertebrate data were available, saltwater invertebrate data were used. If data were not available for invertebrates, fish data were used. BSAFs for organic compounds remained the same between the screening and baseline assessment because average BSAF values were used.

Benthic invertebrate body burdens were derived by multiplying the BSAF directly by the sediment concentration for inorganic contaminants. The benthic invertebrate body burden for organic compounds was calculated as follows, based on McFarland (1998):

$$\text{Benthic invertebrate body burden} = \text{BSAF} \times \frac{\text{sediment concentration mg/kg}}{(\%\ \text{TOC sediment} \times \%\ \text{lipid})}$$

The sediment TOC was assumed to be 1 percent, based on lake data. The average benthic invertebrate percent lipid value was assumed to be 2 percent, based on Lechich (1998), as Onondaga Lake invertebrate lipid data were limited to four chironomid and two (excluding a duplicate) amphipod samples.

Emergent insects were assumed to possess the same COC body burdens as the benthic larvae. All nondetected values were considered as observations at one-half the detection limit, except for groups of compounds as discussed previously. Values for TF_{BSAF} are provided in Table 8-5.

8.2.6.7 Chemical of Concern Concentrations in Water and Sediment/Soil (Incidental Ingestion)

No selection criteria were assumed for drinking water. Pooling all observations of whole water taken from the epilimnion provided an approximate water concentration in Onondaga Lake. The epilimnion data ranged from 0 (surface) to 3 m in depth and can be used by wildlife receptors as a source of drinking water.

Incidental sediment ingestion was confined to the littoral zone (water depth less than or equal to 2 m of standing water) and included all sediment analyses to a depth of 15 cm.

Soil ingestion for the receptors feeding on invertebrate and vertebrate terrestrial prey was based on soils collected from wetlands and the dredge spoils areas and included all observations to depths of 30 and 50 cm, respectively. Data are provided in Appendix I.

8.2.6.8 Food Ingestion Rates

Another component of the exposure assessment is the receptors' food ingestion rates (FIRs). The ingestion rate of an organism is a function of its energy requirements, the energy density (energy content) of its diet, and the efficiency of the organism's energy assimilation from the diet. Body weight estimates for all of the receptors were determined from literature reports. Mean body weights representative of populations indigenous to New York and the northeastern United States were preferred over other locations. Estimates of prey intake rates were based on the bioenergetic scaling relations of Nagy (1987) and expressed as field metabolic rates (FMR) (kcal/day) using data contained in USEPA (1993b). However, this is not the case for the short-tailed shrew and the little brown bat, since very small eutherian mammals were not represented in the data sets used for scaling equations. Therefore, literature sources were used to select FIRs for the shrew and bat, as discussed in the receptor profiles.

Energetic estimates were represented as lognormal distributions and determined as follows:

$$E_B = A(BW)^b = \begin{pmatrix} Birds : 2.601 \times (BW)^{0.640} \\ Mammals(non - herbivores) : 0.6167 \times (BW)^{0.862} \end{pmatrix}$$

$$\mathrm{IR} = \frac{\mathrm{E_B}}{\mathrm{BW} \times \mathrm{E_{Prey}} \times \mathrm{AE}}$$

where:

E_B	=	estimated field metabolic rate for the receptor (kcal/day)
BW	=	receptor body weight (g)
A	=	intercept coefficient of the scaling regression (from Nagy [1987]; kcal/day)
b	=	independent variable coefficient of the scaling regression (from Nagy [1987]; unitless)

IR = estimated mean intake rate (kg dry weight/kg body weight-day)

AE = assimilation efficiency (percent)

E_{prey} = energy content of specific prey type (kcal/g dry weight).

8.2.6.9 Water and Incidental Sediment/Soil Ingestion Rates

To estimate ingestion rates of drinking water, the allometric relation between body size and water ingestion of Calder and Braun (1983) provided below was applied to all receptors as a point estimate.

$$WI(L/day) = \begin{pmatrix} Birds: 0.059 \times (BW)^{0.67} \\ Mammals: 0.099 \times (BW)^{0.90} \end{pmatrix}$$

Information for incidental soil or sediment ingestion is available only for the mallard (Beyer et al., 1994). Modeling for all other receptors was based on closely related species for which incidental soil/sediment ingestion data are available and/or professional judgment. All point estimate ingestion rates used in the analyses are included in Tables 8-6 and 8-7.

8.2.6.10 Chemical of Concern Speciation, Composition, and Bioavailability

Many of the COCs under consideration in this BERA are found in multiple chemical forms in the environment. The form of a chemical affects both its uptake rate and toxicity. Chemical analysis of abiotic media and prey tissue measures the total concentration of chemicals but not necessarily the amount biologically available to the receptors, which may be lower, but never greater. The assumption used in the food-web exposure model is that a specific COC in exposure media is as bioavailable as the form used in the toxicity studies on which the TRV is based. The assumption that COCs in the field are equally as bioavailable as chemicals in laboratory studies is retained in this BERA in the absence of adequate and consistent data on relative bioavailability.

This section discusses the considerations associated with speciation/composition and describes how they were reconciled for key contaminants. All relevant measured COC estimates in all media collected by Honeywell between 1992 and 2000 and by NYSDEC from 1992 to 2000, as described in the Onondaga Lake RI (TAMS, 2002b), were used in the exposure concentration estimates. Non-detected values were included at one-half the detection limit, except in the case of groups of compounds, as described previously.

Mercury

Organomercurial species, such as methylmercury, are more toxic to wildlife than inorganic forms of mercury. Mercury methylation occurs in aquatic habitats and in wetland habitats. Methylmercury, unlike inorganic and metallic mercury, is highly bioavailable and tends to bioaccumulate. In the higher-trophic-level aquatic prey, such as fish, methylmercury will concentrate to where it may comprise up to 95 percent of the total mercury load for an individual fish. Therefore, receptors dependent on fish as prey were assessed based on exposure to methylmercury.

Results of the benthic invertebrate mercury and methylmercury analyses confirmed that methylmercury also accounts for a significant portion of mercury in invertebrates. Methylmercury averaged about 49 percent of the total mercury in amphipods, 16 percent of the total mercury in chironomids, 5 percent of the total mercury in oligochaetes, 53 percent of the total mercury in *Daphnia*, and 64 percent of the total mercury in other zooplankton for an average of 37 percent methylmercury in lake invertebrates (weighting all groups equally). Hence, for receptors reliant on benthic macroinvertebrates as prey, 37 percent of all mercury was assumed to be methylmercury and 63 percent to be inorganic mercury.

One percent of mercury in wetland areas was considered to be methylmercury, as discussed in Chapter 6, Section 6.3.1.1 of this report.

Total PCBs

PCBs were analyzed as specific Aroclors. Because Aroclors are themselves mixtures of PCB congeners, their composition may change over time. The risk from Aroclors was evaluated based on total PCB concentrations and defined as the sum of all Aroclors (within each sample) measured in the respective media.

Aroclor 1254/1260 (combined) was the predominant form of Aroclor detected in NYSDEC fish samples. Fish generally metabolize less chlorinated PCBs (mono and di), while retaining trichlorinated and higher congeners. The more highly chlorinated congeners are the most toxic to fish and wildlife. PCBs were only analyzed in three amphipod samples. Based on the limited data available, UFs were used to calculate PCB concentrations in aquatic and soil invertebrates rather than using the three samples to represent total PCB concentrations.

Dioxins/Furans

Dioxins and furans were assessed using the TEQ approach (Eastern Research Group [ERG], 1998). All chlorinated dioxins and furans were converted to TEQs for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) using toxicity equivalence factors (TEFs) specific to either birds or mammals (van den Berg et al., 1998). The constituents were then summed within each sample and compared to the toxicity of 2,3,7,8-TCDD to evaluate risk. PCBs were not included in the TEQ analyses because they were analyzed as individual Aroclors and not as specific PCB congeners, some of which have TEQ factors available.

DDT and Metabolites

The toxicity of DDT, with regard to impacts on reproduction, is directly caused by the metabolite DDE, which is an intermediate in the catabolism of DDT and DDD. Therefore, the exposure of receptors to DDT was determined based on the sum of DDE and all metabolite precursors (DDT and DDD) within each sample in the respective media.

Other Contaminants

Individual contaminants of the following groups of contaminants were summed together within each sample and exposure was based on the sum of constituents. The best available toxicological data for any compound within the group was used to derive TRVs.

- Total dichlorobenzenes.
- Total trichlorobenzenes.
- Total xylenes.
- Total PAHs.
- Total chlordanes.
- Hexachlorocyclohexanes.

All metals selected as COC for wildlife receptors other than mercury (i.e., antimony, arsenic [arsenite], barium, beryllium, cadmium, chromium [chromic], cobalt, lead, nickel, selenium, thallium, vanadium [vanadate], and zinc) were assessed based on the toxicity and bioavailability of the free ion in its most common oxidized form.

8.2.7 Life History Characteristics of Wildlife Receptor Species

Species-specific characteristics that were used in assessing chemical exposure through the food-web models are discussed in the following sections, along with the basis for their selection. Summaries of the avian and mammalian exposure factors can be found in Tables 8-6 and 8-7, respectively.

8.2.7.1 Tree Swallow (*Tachycineta bicolor*)

The tree swallow is a common perching songbird that breeds throughout the northern half of North America, where they are frequently found in association with bodies of water (Secord and McCarty, 1997). They prefer open areas in the vicinity of water, such as river valleys, lakes, marshes, flooded swamps, and beaver ponds in close proximity to decaying trees. However, they also use fields and meadows, if appropriate nesting sites are available and open water is nearby (Andrle and Carroll, 1988). The average weight of a female tree swallow in New York was reported to be 20.6 g (Secord and McCarty, 1997), which was used in this assessment. This weight corresponds closely to the average spring weight reported for female tree swallows of 20.7 g (Robertson et al., 1992).

Tree swallows are insectivores that pursue flying insects on the wing, using abrupt turns, and sometimes converging in large numbers on insect swarms (Robertson et al., 1992). Food samples from tree swallows nesting along the Hudson River, New York consisted of 50 to 98 percent aquatic emergent insects (Secord and McCarty, 1997), but they may supplement their diet with vegetation during cold spells (Robertson et al., 1992). Tree swallows forage at heights of 0 to 50 m or more above ground, over open areas of water or ground that are sheltered from wind where flying insects accumulate. They feed from dawn to dusk, but most intensively between late morning and late afternoon during the breeding season (Robertson et al., 1992). Swallows, especially bank swallows but possibly others, have been observed nesting in the sides of Wastebeds 1 to 8 along Onondaga Lake. As tree swallows are considered to live near the lake, their diet was assumed to consist of 100 percent of aquatic insects.

The FIR (dry-weight basis) was estimated as 0.264 kg/kg-day based on a field metabolic rate of 875 kcal/kg body weight per day (based on Nagy, 1987).The daily drinking water intake rate (WIR) for tree swallows was estimated as 0.21 L/kg-day (based on Calder and Braun, 1983). Minimal contact with sediments is expected during feeding and grooming and, therefore, incidental sediment ingestion was set at zero.

Home-range size varies according to season and geographic area, but is between 0.1 and 0.2 km in New York (McCarty and Winkler, 1999). During the breeding season parents make 10 to 20 trips per hour to feed their nestlings (Quinney, 1986). Tree swallows nest in abandoned, excavated woodpecker holes, natural cavities in standing trees, or artificial nest boxes located in open fields or near water (Robertson et al., 1992). Most nests are spaced 10 to 15 m apart, but occasionally breeding pairs are found as close as 1 to 3 m apart.

Migrating tree swallows arrive in their northern breeding areas from February through April, with most arriving in March. Tree swallows in New York begin defending nest boxes and gathering nest material by late April, and commence egg laying by May (Secord and McCarty, 1997; Andrle and Carroll, 1988). Fall migration to wintering ranges occurs between July and September, with late August being the peak migration time (Robertson et al., 1992). A holdover population of tree swallows has been sighted regularly in the Syracuse area during the Audubon Christmas counts (Cornell University, 2001), indicating that some tree swallows remain in the Onondaga Lake area year-round and, therefore, exposure was set at 365 days per year.

8.2.7.2 Mallard (*Anas platyrhynchos*)

The mallard is one of the most common species of waterfowl in New York State (Bull, 1998). Mallards are dabbling ducks that forage by sifting through sediment in search of aquatic plants, seeds, and invertebrates (USEPA, 1993b). Although mallards are primarily herbivorous, females may switch to a diet with a larger component of invertebrates in spring in preparation for molting and egg-laying (Swanson et al., 1985; Heitmeyer, 1988). Ducklings also consume aquatic invertebrates almost exclusively during their period of rapid growth (Chura, 1961). Based on these studies, a female mallard was assumed to have a diet comprised of 50 percent aquatic invertebrates and 50 percent aquatic plants.

The average weight of female mallard in North America is 1,043 g, based on the weights of over 3,000 birds (Nelson and Martin, 1953). This weight corresponds to a field metabolic rate of 213 kcal/kg body weight per day (based on Nagy, 1987) and a consumption of 0.101 kg dry weight/kg body weight per day.

The drinking WIR was estimated at 0.058 L/kg body weight-day using Calder and Braun (1983). The sediment ingestion rate (SIR) was assumed to be 3.3 percent, based on the analyses of mallard scat (Beyer et al., 1994).

Female mallard home-range sizes vary from an average of 111 to 540 hectares, depending on habitat features such as size and distribution of available aquatic habitats (Dwyer et al., 1979; Kirby et al., 1985). Other factors shown to affect foraging range are gender, reproductive status, and population density (Dwyer et al. 1979; Kirby et al. 1985). The Onondaga Lake area was considered large enough to support a mallard population.

Migratory and resident mallards are found throughout New York State (Andrle and Carroll, 1988). Substantial numbers of mallards have been regularly documented in the Onondaga Lake area during the annual Christmas bird count (Cornell University, 2001). Based on these observations, mallards were considered to be year-round residents of Onondaga Lake.

8.2.7.3 Belted Kingfisher (*Ceryle alcyon*)

The belted kingfisher is found throughout much of North America (Bent, 1940). Although it typically inhabits areas around lakes, ponds, wooded creeks, rivers, bays, and estuaries, it is found in every ecozone in New York State (Andrle and Carroll, 1988). The belted kingfisher is an aquatic feeder and requires clear waters in order to see their prey (Davis, 1982; Salyer and Lagler, 1946). Kingfishers perch on a tree limb over a body of water while searching for prey and fish mainly at the surface of the water. The average body weight of an adult belted kingfisher selected for this assessment was 136 g based on a Pennsylvania population (Brooks and Davis, 1987).

Fish are the predominant prey of the belted kingfisher, as its name implies (Bent, 1940; USEPA, 1993b). However, diets can vary with prey availability and kingfishers may supplement their diets with aquatic macroinvertebrates, terrestrial prey, and/or plant material (Alexander, 1977). Fish are assumed to represent 100 percent of the total kingfisher diet in this assessment. Prey typically collected by the belted kingfisher range between 4 and 14 cm (Davis, 1982; Brooks and Davis, 1987), although they may consume fish up to 18 cm (Salyer and Lagler, 1946). Kingfishers appear to take prey in proportion to the relative abundance of each size (Davis, 1982). Fish less than 18 cm in length were used to model prey contaminant concentrations for the belted kingfisher.

The allometric equation of Nagy (1987) was used to estimate the bioenergetics for the belted kingfisher. Based on a daily metabolic field rate of 444 kcal/kg body weight per day, an average intake rate of 0.137 kg dry weight/kg body weight per day was determined. The drinking water intake was estimated at 0.114 L/kg body weight-day based on the algorithm of Calder and Braun (1983).

Incidental sediment ingestion during nest building and grooming was assumed to be 1 percent of total prey intake to account for soil ingestion during nest construction and nesting, as belted kingfishers construct their nests by excavating tunnels in embankments (Levine, 1988). Although the kingfisher hunts almost exclusively within the pelagic zone, both the male and female dig the nesting burrow, using their bills as probes and their feet as shovels (Andrle and Carroll, 1988).

Home range is typically defined by length of shoreline defended by mated pairs (breeding territory) and feeding areas defended by solitary adults (non-breeding). Generally, breeding pairs defend a larger habitat than solitary individuals, although considerable overlap in size occurs. Kingfishers establish and defend summer territories for nesting and feeding. The foraging range of the kingfisher was reported to average between 0.4 and 2.2 km (Davis, 1982; Brooks and Davis, 1987). The Onondaga Lake foraging range was assumed to be 1 km based on the breeding Ohio population studied in Davis (1982). Resident kingfishers were considered to rely solely on the lake as their foraging habitat.

The timing and extent of migration appears to be related to the severity of the weather (Davis, 1982). The belted kingfisher is a hardy bird, and it remains as far north in fall and winter if it can find open water in which to catch a sufficient number of fish (Bent, 1940). Audubon Christmas counts in the Syracuse area have consistently recorded the belted kingfisher (Cornell University, 2001); therefore a year-round residency time was assumed. In addition, full exposure is considered appropriate because belted kingfishers are exposed to lake contaminants during sensitive reproduction and growth periods when their vulnerability is greatest (i.e., April to August).

8.2.7.4 Great Blue Heron (*Ardea herodias*)

The great blue heron is a wading bird that occurs in a variety of freshwater and marine habitats and breeds throughout much of North America (Bent, 1926). It is the largest member of the heron family in North America. Great blue herons may inhabit lakes, rivers, brackish marshes, lagoons, coastal wetlands, tidal flats, and sandbars, as well as occasional wet meadows and pastures (USEPA, 1993b). An average body weight for the female great blue heron of about 2,200 g was selected based on Dunning (1993).

The principal food of the great blue heron is fish of various kinds, but amphibians (e.g., frogs), snakes, small mammals, and aquatic and terrestrial invertebrates are also taken on occasion (Bent, 1926; Palmer, 1962). The great blue heron fishes by still hunting and stalking (Bent, 1926). Still hunting is the commonest method, where the heron stand motionless waiting for prey (primarily fish), which it captures striking swiftly with its bill (Eckert and Karalus, 1983). Great blue herons may also slowly wade in shallow water until it drives a fish out from a hiding place (Environment Canada, 2002). Fish make up 90 to 98 percent of the diet, with the rest consisting of crustaceans, insects, amphibians, reptiles, birds, and small mammals (Alexander, 1977; USEPA, 1993b). In this analysis, fish were assumed to comprise 100 percent of the dietary intake.

Great blue herons mainly eat fish 3 to 33 cm in length (Alexander, 1977), but may consume fish as large as 60 cm (Eckert and Karalus, 1983). Krebs (1974) found that smaller prey were selected more frequently because of greater abundance and less handling time. Although a greater number of small fish are eaten, the majority of the diet by weight consists of large fish. A proportion of two-thirds large fish (greater than 18 cm) and one-third small fish (less than or equal to 18 cm) was used to estimate fish consumption of the heron. Based on these assumptions and a field metabolic rate of 163 kcal/kg body weight per day, using the bioenergetic algorithm of Nagy (1987), the daily FIR was estimated to be 0.0445 kg/kg (dry-weight basis).

The drinking WIR was estimated at 0.045 L/kg body weight-day based on the algorithm of Calder and Braun (1983). Data were not available on incidental SIR, which was assumed to be 1 percent, based on fishing techniques.

The average foraging ranges for the great blue heron in South Dakota ranged from an average of 3.1 km to a maximum distance flown of 24 km (Dowd and Flake, 1985). Foraging ranges of herons overlapped with mean densities of 2.3 birds/km and 3.6 birds/km observed at two separate locations (Dowd and Flake, 1985). An average home range of 3.1 km was assumed for this assessment. Based on the home range, the range overlap of individual birds, and the 16 km shoreline of Onondaga Lake, it was assumed that Onondaga Lake could support a small great blue heron population.

In New York State, the great blue heron can be both a seasonal migrant or a resident species throughout the year as long as open water persists (Bull, 1998). Results of the Audubon Christmas Bird Count show that the great blue heron is a regular winter resident in the Onondaga Lake area (Cornell University, 2001). Migrations in the northeast are highly dependent upon the severity of the winter season, primarily the degree of ice cover on feeding waters. During severe conditions (i.e., persistent cold and continuous ice cover), northeast populations will migrant south to portions of the Carolinas and Virginia. Fall migration in the Onondaga Lake population remains unclear given the tendency of this species to linger or reside in summer grounds during the winter period, and hence herons were assumed to be year-round residents.

8.2.7.5 Osprey (*Pandion haliaetus*)

The osprey is a large, powerful raptor that resembles an eagle, but its narrow wings are markedly angled when outspread and the structure of its feet and claws is so peculiar that it has been placed in a separate subfamily, the Pandioninae, of which it is the sole representative (Environment Canada, 2002). It is distributed throughout North America and found near both freshwater and saltwater environments. The average weight of an adult female is 1,568 g, while the males are slightly smaller averaging 1,403 g (Brown and Amadon, 1968).

Osprey are almost always associated with water, usually a river, lake, or the sea coast, although to reach some of these areas and during migration they may pass over large land areas (Brown and Amadon, 1968). Ospreys are skilled fishers and feed almost entirely on fish, although on occasion they may take other prey including birds (possibly wounded), frogs, and crustaceans (Brown and Amadon, 1968). On sighting prey, they hover briefly at a height of 10 to 30 m until the fish is in a suitable position. It then dives into the water, usually reappearing with a fish in its claws, which may be as large as two kilograms (Brown and Amadon, 1968; Environment Canada, 2002).

Ospreys fishing near a reservoir in Idaho consumed fish up to 41 cm in length, with the majority of prey 11 to 30 cm in length (Van Daele and Van Daele, 1982). The mean weight of fish taken by ospreys in

Chesapeake Bay was 237 g in 1975 and 157 g in 1985 (McLean, 1991). Based on these studies, the Onondaga Lake osprey was assumed to assume have 10 percent of its fish consumption made up of fish less than or equal to 18 cm and 90 percent consisting of fish greater than 18 cm.

Based on a diet consisting entirely of fish and a field metabolic rate of 184 kcal/kg body weight per day, using the bioenergetic algorithm of Nagy (1987), the daily FIR was estimated to be 0.048 kg/kg (dry weight basis).

A drinking WIR of 0.051 L/kg body weight-day was estimated, based on free-living metabolic rate (Calder and Braun, 1983). No significant sediment ingestion was assumed for this species, as it has minimal contact with sediments during feeding and nesting.

Ospreys have been observed nesting in Clark Marsh (5.7 km from Onondaga Lake) on an annual basis (Clark, pers. comm., 2000,) and have been observed near the lake (Tango, 1993). The average foraging radius for ospreys ranges from 1.7 to 10 km (USEPA, 1993b). Therefore, Onondaga Lake was assumed to make up the majority of the source of food for some osprey.

Northern populations of ospreys migrate to warmer areas in the winter. Ospreys depart for their wintering grounds around the end of September and the spring migration reaches Onondaga County the first week of April (Purcell, 2001). This migratory pattern yields a typical residency time in New York of about half the year (183 days/year), but the osprey feeds at Onondaga Lake during sensitive periods of growth and reproduction (i.e., April to September). Therefore, a year-round residency time of 365 days per year was used to calculate osprey exposure.

8.2.7.6 Red-Tailed Hawk (*Buteo jamaicensis*)

The red-tailed hawk is one of the most widespread birds of prey in North America, with breeding populations distributed throughout most of the continent (Preston and Beane, 1993). They are highly mobile predators that often inhabit heterogeneous habitats (Preston, 1990). Adult females average 1,224 g, while males are smaller, averaging 1,028 g (Dunning, 1993).

The red-tailed hawk is classified as an avian carnivore with a diet consisting primarily of small mammals (about 70 percent), birds (about 18 percent), and reptiles (about 11 percent), with occasional amphibians, fish, and arthropods (Marti and Kochert, 1995). Its diet varies according to prey availability. For the purpose of this BERA, the potential risk due to exposure to COCs was modeled based on 100 percent small mammal consumption. Using the bioenergetic algorithm of Nagy (1987), a daily field metabolic rate of 246 kcal/kg body weight per day was estimated. This estimate yielded an FIR of 0.052 kg/kg-day dry weight for this species. Drinking WIR was estimated at 0.055 L/kg body weight-day, based on the algorithm of Calder and Braun (1983). An SIR of 1 percent was assumed based on professional judgment, because while some soil attached to prey may be ingested, the amount is assumed to be minimal.

The home range of the red-tailed hawk varies depending on topography, food availability, human activity, and season (Preston and Beane 1993). The average territory for the red-tailed hawk ranges from 60 to 1,770 hectares (USEPA, 1993b). Sample and Suter (1994) recommend using a home range of 233 hectares based on a study in Oregon by Janes (1984), which equals 2.3 sq km. This species has been noted to nest in the vicinity of Onondaga Lake during 1980 through 1985 (as reported by the NYSDEC's Breeding Bird Atlas Project). In 1981 and 1982, the New York State Breeding Bird Atlas Project noted nests with young and in 1983, recently fledged young were spotted near Onondaga Lake and its tributaries. Based on the area surrounding Onondaga Lake and observations, a small red-tailed hawk population is assumed to feed solely in the lake area. One resident population covering the entire area of the lake was modeled for this BERA.

Many red-tailed hawks breeding in northern regions migrate south. However, even in the harshest winters with extensive snow cover, some birds remain near their breeding territory year-round (Preston and Beane, 1993). The red-tailed hawk has been regularly spotted during the Audubon Christmas count in the Onondaga Lake area (Cornell University, 2001). Therefore, some individuals were assumed to have year-round residency (365 days per year) in the Onondaga Lake area.

8.2.7.7 Little Brown Bat (*Myotis lucifugus*)

The little brown bat is common throughout North America, including most of the United States and Canada. This insectivorous species is indigenous to New York State where it is considered a non-game species and is regulated by NYSDEC. Bats collected at the end of August at Ironville, New York had average weights of 8.8 g for females and 7.2 g for males (Davis and Hitchcock, 1965). These bats were collected at about their maximum weight, since July and August are spent in heavy feeding as bats build up their fat reserves before hibernation. The mean weight of female bats in the New York State Museum collection was 7.1 g, which was used to represent Onondaga Lake bats. These weights agree with the average adult weight of 6 to 8 g for little brown bats studied near Ithaca, NY (Wimbatt, 1945) and in New Jersey (McManus and Esher, 1971).

Little brown bats are nocturnal mammals that feed on insects primarily near bodies of water (Barbour and Davis, 1969). Foraging flights of little brown bats begin at dusk and last for 1.5 to 3 hours, with a second feeding period lasting for more variable periods of time, until dawn (Anthony et al., 1981). Fecal analysis revealed that little brown bats consume varied insect taxa, including Diptera, Lepidoptera, Coleoptera, Ephemeroptera, Hymenoptera, Trichoptera, and Neuroptera, typically 3 to 10 mm in length (Anthony and Kunz, 1977). Belwood and Fenton (1976) reported that according to fecal analysis, aquatic insects, primarily Diptera (chironomids) and Tricoptera (caddis flies), constituted about of the 95 percent of the adult diet at a site in northern New York, although Buchler (1976) observed that Ephemeroptera (mayflies) comprised the majority of the diet of his study population.

The amount of prey ingested during feeding varies by sex, age, and reproductive state. On average, pregnant bats ingested 2.5 g of prey, lactating females ingested 3.7 g, and juveniles ingested 1.8 g per feeding flight (Buchler, 1976). Digestion of ingested prey begins after the stomach is full and the bat has returned to its colony. Transit time in the gut is rapid, and complete digestion and excretion of one stomach volume can take less than an hour for an active individual, allowing bats to fill their stomach two or more times each feeding period (Buchler, 1976). Little brown bats may consume between 30 and 100 percent of their body weight each night (Hoffman, 1999; Environment Canada, 2000; Snyder, 2002).

No field metabolic rate measurements have been made on very small, active eutherians. Therefore, information contained in the literature was used to estimate prey consumption rates. A consumption rate of 25 percent of the body weight (1.8 g/day) was selected to represent an average feeding rate for female bats, considering both their active and hibernating periods. The wet weight consumption rate was converted to dry weight using the a conversion rate of 1 kg dry weight to 4.5 kg wet weight based on the average wet weight to dry weight ratio for aquatic invertebrates for studies listed in Wildlife Exposure Factors Handbook (USEPA, 1993b). This value is within the range of conversion factors provided by Peters (1983). Food consumption on a dry-weight basis was therefore estimated as 0.102 kg dw/kg body weight per day based on the energy content of insect larvae taken from USEPA (1993b).

The WIR was estimated to be 0.162 L/kg body weight per day (Calder and Braun 1983). No SIR was assumed for this species, since bats capture smaller insect prey directly with the mouth in flight and use their body, tail, and wings to cup and direct larger prey into the mouth. All insect prey is masticated and devoured in flight.

A home range of 0.1 km was selected for the little brown bat based on observations of the distance traveled by a colony in New York for nightly feeding by Buchler (1976).

In response to colder temperatures and diminishing prey, little brown bats move from summer roosting and maternity colonies to winter hibernacula (i.e., hibernating shelters). Seasonal movements occur before and after hibernation. While not truly migration in definition, this movement results in temporary displacement of little brown bat populations from summer refuges/feeding areas and dispersal to other summer and winter refuges. The distance traveled by bats from New York and New England populations ranged from 8.7 to 105 km between summer and winter locations (Griffin, 1945; Davis and Hitchcock, 1965). In New York little brown bats were found to return to winter hibernacula from September to October (Davis and Hitchcock, 1965), with females leaving the hibernaculum earlier (April to mid-May) than males (mid-May to early June) to disperse or move to summer colonies. Although the little brown bat hibernates part of the year and may move from out of the Onondaga Lake vicinity, all food sources during the year (i.e., active feeding time and fat reserves used during hibernation) are assumed to be derived from Onondaga Lake. Reproduction and growth (the most sensitive time periods) also occur when the little brown bat is active at Onondaga Lake. Therefore, the little brown bat was treated as a year-round resident of Onondaga Lake and no temporal modifying factor was applied.

8.2.7.8 Short-Tailed Shrew (*Blarina brevicauda*)

The short-tailed shrew (*Blarina brevicauda*) is a small insectivorous mammal that ranges throughout the United States (George et al., 1986). Short-tailed shrews range from about 9.5 to 13 cm in length and weigh

12.5 to 22.5 g (Guilday, 1957). An average body weight of 15 g was used based on the average shrew weight in New Hampshire (Schlesinger and Potter, 1974).

Shrews are mainly insectivorous and carnivorous, but some eat seeds, nut meats, and probably other plant material (Nowak, 1997). Analyses of stomach contents of New York State shrews show that earthworms comprise the majority of the short-tailed shrew diet with slugs, snails, insect and miscellaneous animals contributing most of the remainder (Whitaker and Ferraro, 1963). For this assessment, the diet of the shrew was assumed to consist of 100 percent terrestrial invertebrates and modeled contaminant concentrations in earthworms were used to estimate body burdens of contaminants in prey.

The bioenergetic algorithm of Nagy (1987), does not include data for very small, very active eutherian mammals, such as the shrew. Since the field metabolic is strongly correlated with body size, it was considered inappropriate to use Nagy's equation to calculate a metabolic rate for shrews, as those data were not used to develop the equation. Shrews feed frequently and may consume more than their total body weight in food over a 24-hour period (Schmidt, 1994). In the laboratory, food consumption rates ranged from an average of 8 to 10 g/day (Buckner, 1964 and Barrett and Stueck, 1976; both cited in Sample and Suter, 1994). The higher average consumption rate of 10 g/day (two-thirds of the average body weight) was selected since field metabolic rates are likely to be higher than laboratory rates. This equals a daily consumption rate of 0.157 kg/kg-day on a dry-weight basis. Consumption rates do not consider increased food requirements in winter (about 40 percent greater) to maintain body temperatures in colder weather (Randolph, 1973), and therefore this consumption rate may underestimate Onondaga Lake food consumption.

The shrew has a high rate of evaporative water loss and an estimated WIR of 0.151 L/kg-day was calculated based on Calder and Braun (1983), although this equation also has more uncertainty associated with predictions for small sizes. Incidental soil ingestion was assumed to be 13 percent of food consumption (Talmage and Walton, 1993, as cited in Sample and Suter, 1994).

Short-tailed shrews are found in nearly all land habitats(Nowak, 1997). They construct runways in leaves, plant debris, snow, or the ground. Runways are usually in the top 10 cm of soil, but can be as deep as 50 cm (USEPA, 1993b). Home ranges for New York State shrews in the winter average 0.05 hectares, with maximum ranges of 0.10 to 0.22 hectares (Platt, 1976).

Shrews are active year-round and can be seen by day or night (Nowak, 1997). Five resident shrew year-round populations were modeled for this BERA, one living in each of the four wetland areas and the other in the dredge spoils area.

8.2.7.9 Mink (*Mustela vison*)

Mink are distributed throughout all of New York State and most of the United States and Canada (NYSDEC, 2000). They occupy wetland habitats including streams, lakes, rivers, and freshwater and saltwater wetlands. They prefer wetlands and riparian habitat with irregular shorelines, good cover (i.e., woods and shrub), and suitable den sites (Linscombe, et al. 1982; Allen, 1984), but in Sweden are most abundant near eutrophic lakes (Eagle and Whitman, 1987). Regardless of the type of habitat used, mink dens are always associated with water and typically are 5 to 100 m from a water body and mink can use several den sites within their home range. The most widely used den sites are bank burrows of other animals, particularly muskrats. Mink are reasonably tolerant of human disturbance/development as long as prey abundance is not affected (Allen, 1984).

Mink exhibit a pronounce sexual dimorphism in size with male 1.4 to 1.8 times heavier than females (Eagle and Whitman, 1987). An average body weight for mink of 600 g was used based on the average adult female weight provided in Mitchell (1961).

Mink are nocturnal in habit and opportunistic in diet. Although not totally restricted to wetland or wetland-associated habitats, the mink is dependent on aquatic organisms for much of the year (Allen, 1984). As a carnivore in an aquatic habitat, mink may concentrate environmental pollutants (Eagle and Whitman, 1987). They actively seek prey within their home range, and their diet varies according to season, prey availability, and habitat type (Allen, 1984). Mink feed primarily on small aquatic and terrestrial animals, although they can feed upon prey items larger than themselves, such as waterfowl and muskrats (Sealander, 1943). Common prey items include fish, frogs, crayfish, salamanders, clams, insects, muskrats, voles, and rabbits (USEPA, 1993b). Hunting in aquatic habitats occurs in shallow, nearshore areas where aquatic prey is captured and then moved to the shore prior to consumption (Doutt et al., 1977).

A Michigan riverine mink population fed mainly on fish (85 percent), catching fish ranging from 5 to 18 cm (Alexander, 1977). Fish, shellfish, and crayfish were the major food items of mink inhabiting coastal habitats of Alaska and British Columbia (Allen, 1984). A study in Idaho (Melquist et al., 1981) found fish occurring 59 percent of the time in the mink diet with unidentified cyprinids, ranging in length from 7 to 12 cm comprising the major type of fish eaten. A study of summer mink scat in Montezuma Marsh, a wetland in the Finger Lakes region of NY, found the diet to consist primarily of mammals (43 percent mammals), fish (27 percent), aquatic invertebrates (14 percent), and birds (9 percent) (Hamilton, 1940). The most abundant forage fish in Montezuma Marsh, the golden shiner (*Notemigonus crysoleucas*), comprised the greatest proportion of fish in the mink diet. Fish consumed from Montezuma Marsh were generally 8 to 11 cm, while mink belonging to a Montana riverine population feed mainly on brook stickleback *(Culaea incostans)* from about 4 to 6 cm (Gilbert and Nancekivell, 1982). Fish are also a common food item of mink during winter months (NYSDEC, 2000a).

Onondaga Lake mink were assumed to consume a diet consisting of 35 percent fish, 15 percent aquatic invertebrates, and 50 percent other food sources (e.g., mammals, waterfowl, amphibians). This dietary composition was selected to represent year-round exposure at Onondaga Lake. Small mammals were selected to represent "other" food sources, as no body burden data were available or modeled for birds or amphibians. Mink were assumed to feed on fish 18 cm or less in length. All fish in this size range were used to estimate fish contaminant concentrations. However, as the mink is an opportunistic feeder, prey selection often depends primarily on the abundance of fish or other prey species and secondarily on the size.

An average field metabolic rate for female mink was estimated as 255 kcal/kg body weight per day based on the algorithm of Nagy (1987). This estimate yields an intake rate of 0.064 kg dry weight/kg body weight-day, based on an assumption of 35 percent fish, 15 percent aquatic invertebrates (e.g., crayfish, beetles), and 50 percent mammals. This value is slightly higher than the daily feed consumption of caged female mink (0.05 kg dry weight/kg body weight-day; Bleavins and Aulerich, 1981), but mink in the wild are expected to have higher energy requirements than caged mink. The daily drinking WIR for the mink was estimated as 0.104 L/kg-day, using the allometric equation of Calder and Braun (1983).

Mink incidentally ingest a small quantity of vegetation and soil while feeding (Alexander, 1977; Sealander, 1943). Based upon the observations of soil and vegetation in mink stomachs (Hamilton, 1940), the percent ingestion of sediment during feeding and grooming was assumed to be approximately one percent of the diet.

Home ranges for the males are normally larger than those of females, particularly during the breeding season (Eagle and Whitman, 1987). During the breeding season, male home ranges may overlap those of several females. However, same-sex ranges never overlap (Eagle and Whitman, 1987). A female's home range, which includes both dens and foraging areas around waterways, may occupy from 1 to 2.8 km of shoreline, depending on food availability, age, gender, and season (Gerrell, 1970). Female mink have the smallest and most well-defined home range, while male ranges tend to be larger and less clearly defined. Mitchell (1961) reported home range sizes of 7.8 and 20.4 hectares for two female mink. A mean home range of 1.85 km of shoreline was selected for this assessment, based on Gerrell (1970). During daily activity periods mink move back and forth in a restricted core area that typically is less than 300 m in shoreline length (Gerrell, 1970, as cited in Allen, 1984). Dens have been reported to lie between 5 and 100 m from water (Melquist et al., 1981). This limited ranging behavior may preclude minks resident to Onondaga Lake from using any other major water body as a food source. Therefore, it was assumed in the food-web model that the mink's diet was derived entirely from the study area for the entire year.

Mink are active year-round and do not hibernate (Doutt et al., 1977; Alexander, 1977). They occupy and defend a resident territory throughout the year and do not migrate, with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations (Allen, 1984). Populations within the study area of Onondaga Lake are year-round residents.

8.2.7.10 River Otter (*Lutra canadensis*)

The river otter is one of the larger members of the Mustelidae family. It is found throughout most of North America and is indigenous to New York State. It is morphologically adapted for land and water, and feeds almost exclusively on aquatic prey. Females are smaller than males with weights ranging from 5 to 15 kg at sexual maturity (Melquist and Dronkert, 1987). In New York State, the average weight is about 5.45 kg (NYSDEC, 2000a), which was selected for use in this assessment.

Fish comprise the majority of otter prey, but otter also commonly feed on crayfish, with aquatic invertebrates, amphibians, birds, mammals, and blueberries contributing a smaller percentage of the diet (e.g., Hamilton, 1961; Knudsen and Hale, 1968; Serfass et al. 1990; Sheldon and Toll, 1964; Toweill, 1974). A diet of 90 percent fish (Newell et al., 1987) and 10 percent aquatic invertebrates was used to estimate dietary exposure of otter to contaminants. Prey for the river otter is generally fish with a reported size range between 2 and 50 cm (Melquist and Dronkert, 1987). Prey availability and catchability influence dietary composition. Common fish eaten by otter include both forage fish and game and pan fish, depending on the area (Tumilson and Shalaway, 1985). Few studies provided relative proportions of size distribution of fish consumed by otter, although work by Toweill (1974) clearly showed a preference for larger fish. Limited data provided in Alexander (1977) indicate that otter prefer feeding on larger fish, and hence a diet of two-thirds (67 percent) fish greater than 18 cm in length and one-third (33 percent) less than or equal to 18 cm in length was assumed.

An average field metabolic rate for a river otter of 5.45 kg was estimated as 188 kcal/kg body weight per day based on Nagy (1987). This estimate yields an intake rate of 0.044 kg dry weight/kg body weight-day, for a diet composed of 90 percent fish and 10 percent aquatic invertebrates. The daily drinking WIR for the river otter was estimated as 0.084 L/kg-day by using the allometric equation of Calder and Braun (1983). Incidental vegetation material occur commonly in the digestive tracts of otters (Toweill, 1974), and therefore the assumption of incidental soil ingestion was set at 1 percent of total daily food intake, as was done for the mink.

The shape and size of the otter home range varies by habitat type. Home ranges have been documented to range from 1 to 78 km (Melquist and Dronkert, 1987). Spinola et al. (undated) monitored otters released along the Genessee River in western New York. They found dispersal distances ranging from 1.5 to 22.5 km, with an average of 10 km for all otters and 9 km for female otters. Home ranges of otters have been shown to overlap extensively, both within and between genders (Erickson and McCullough, 1987). Average densities of otter range from one every 2 to 3 km (Erlinge 1967, 1968, as cited in Nowak, 1997) to one per 3.9 km of waterway (Melquist and Hornocker, 1983). As the shoreline of Onondaga Lake is approximately 18 km excluding the shoreline areas associated with tributaries (see Chapter 3, Section 3.2.1), the Onondaga Lake shoreline was considered adequate to support a small river otter population.

River otter are active year-round and do not hibernate (Doutt et al., 1977). River otters occupy and defend a resident territory throughout the year and do not migrate with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations, and are therefore considered year-round residents.

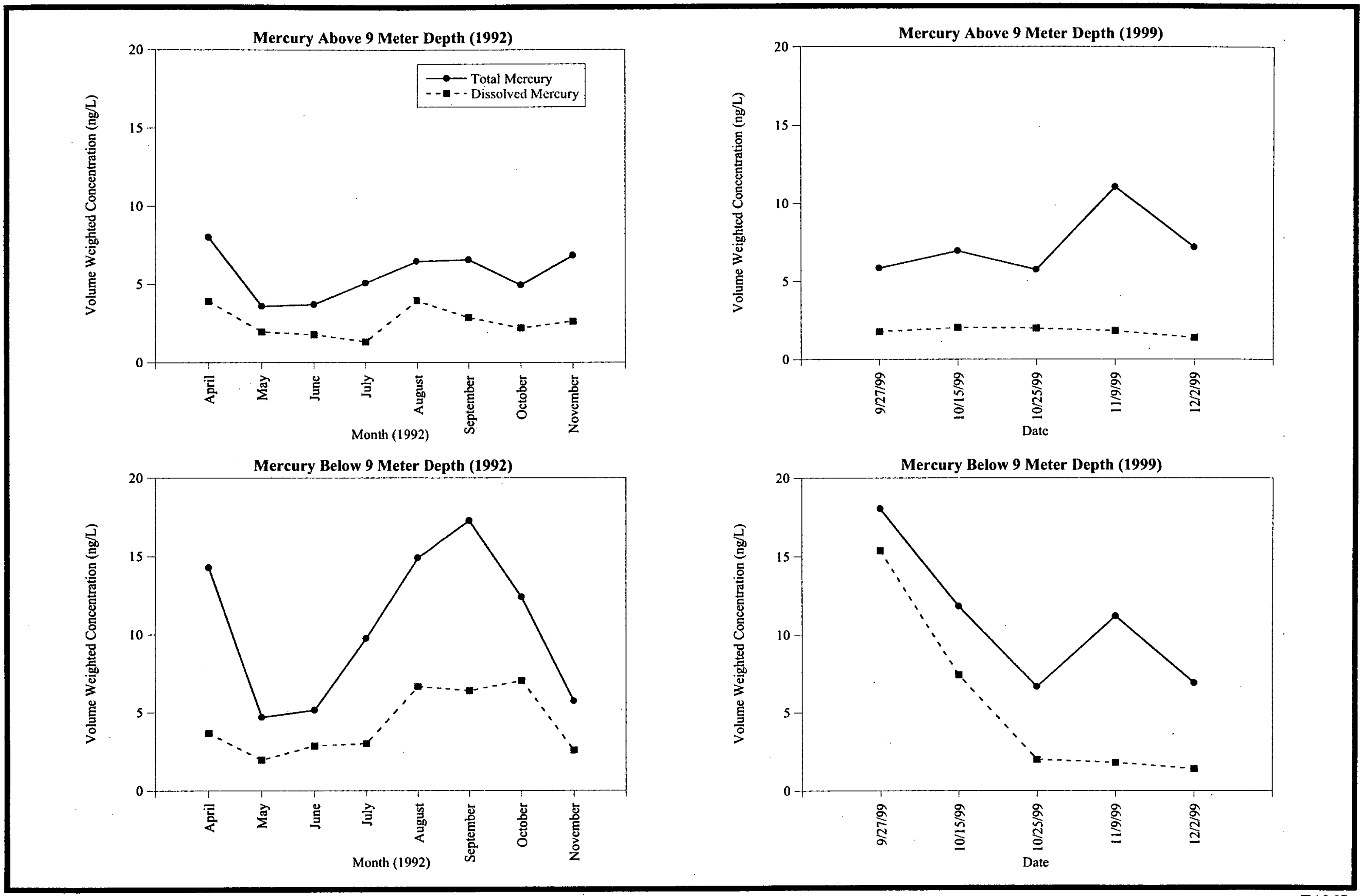
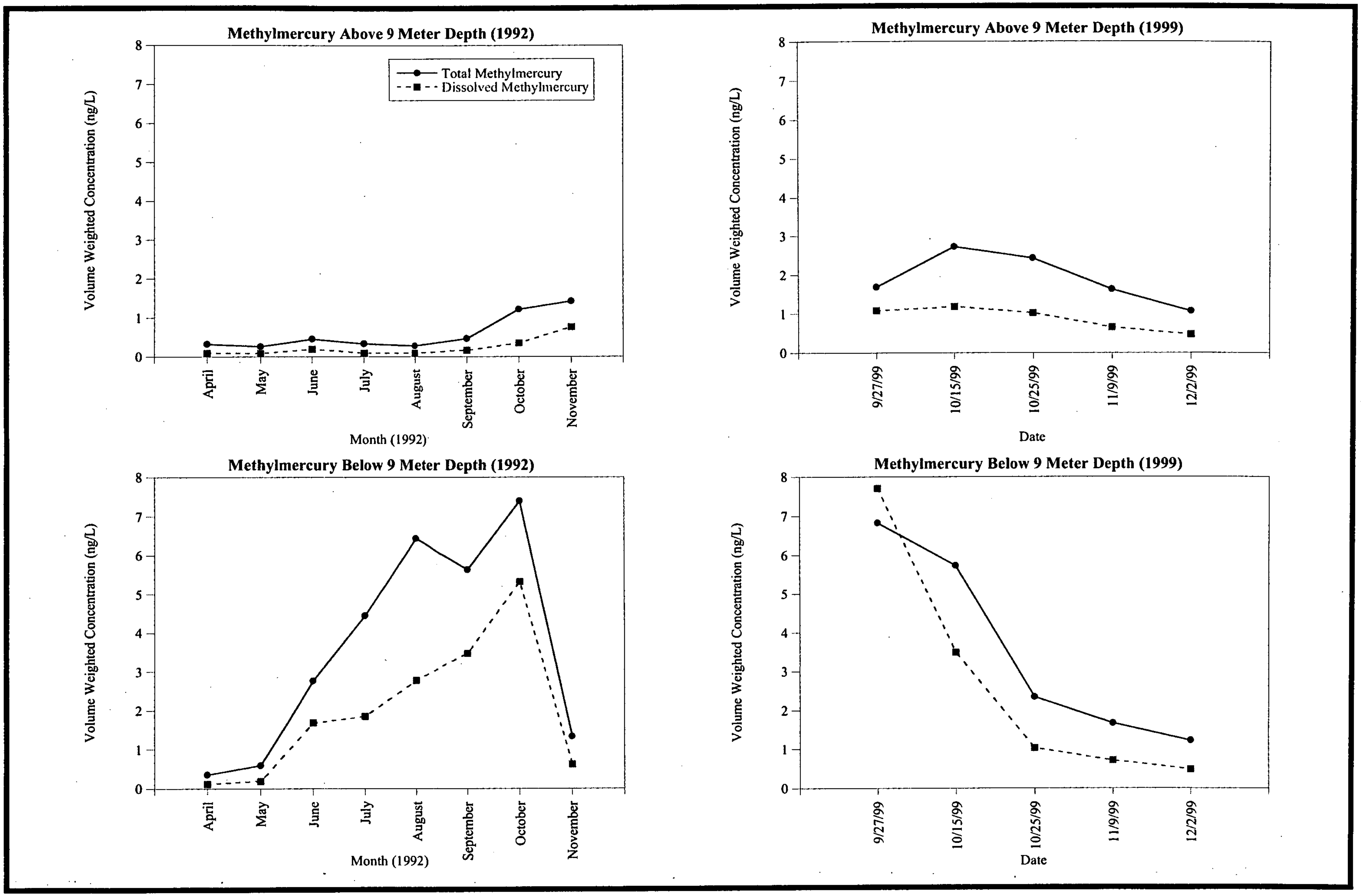


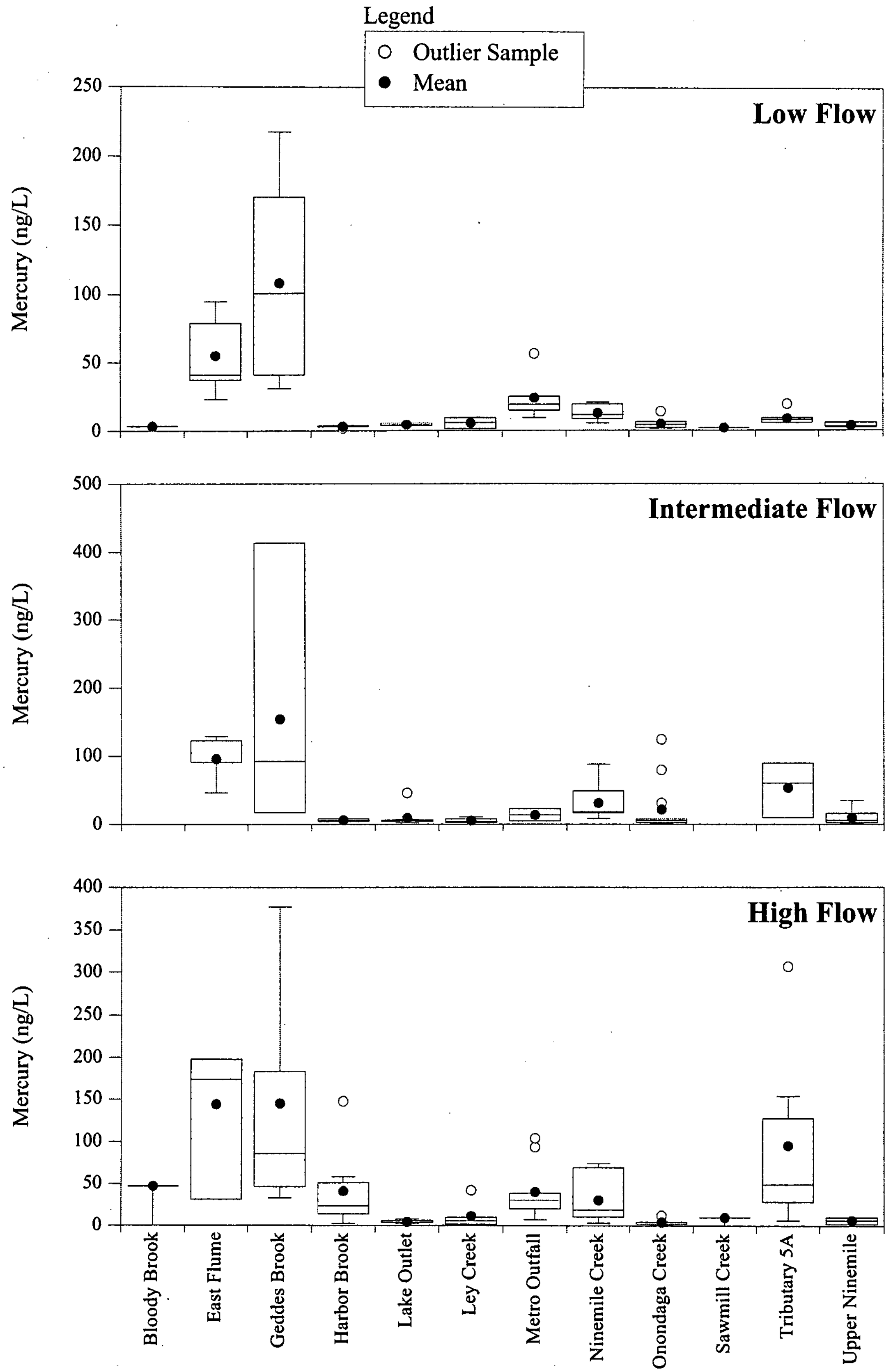
Figure 8-1
Mercury Concentrations in Surface Water of Onondaga Lake

TAMS



TAMS

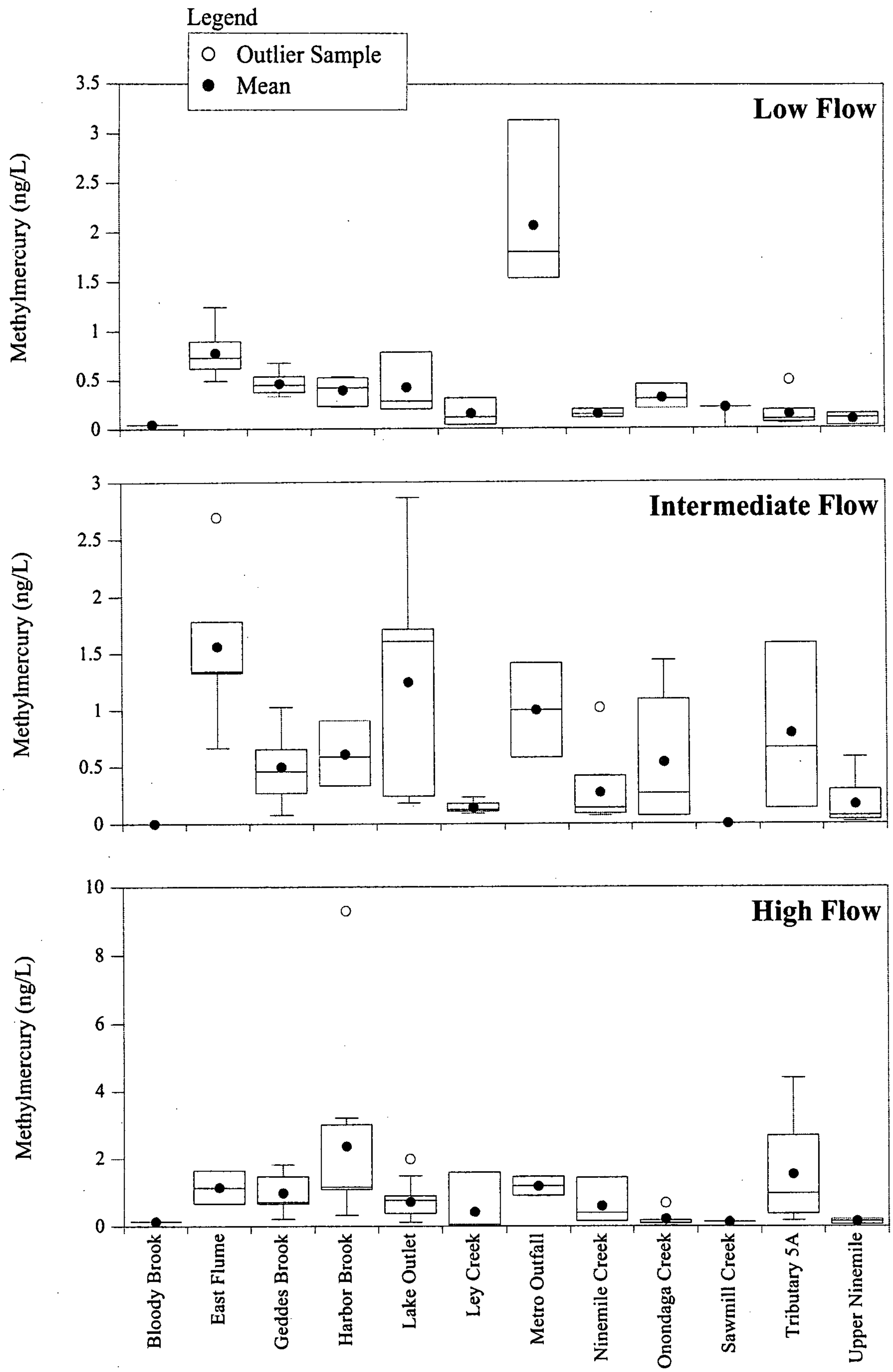
Figure 8-2
Methylmercury Concentrations in Surface Water of Onondaga Lake



Notes:
1. Half the detection limits were used for non-detects.

TAMS

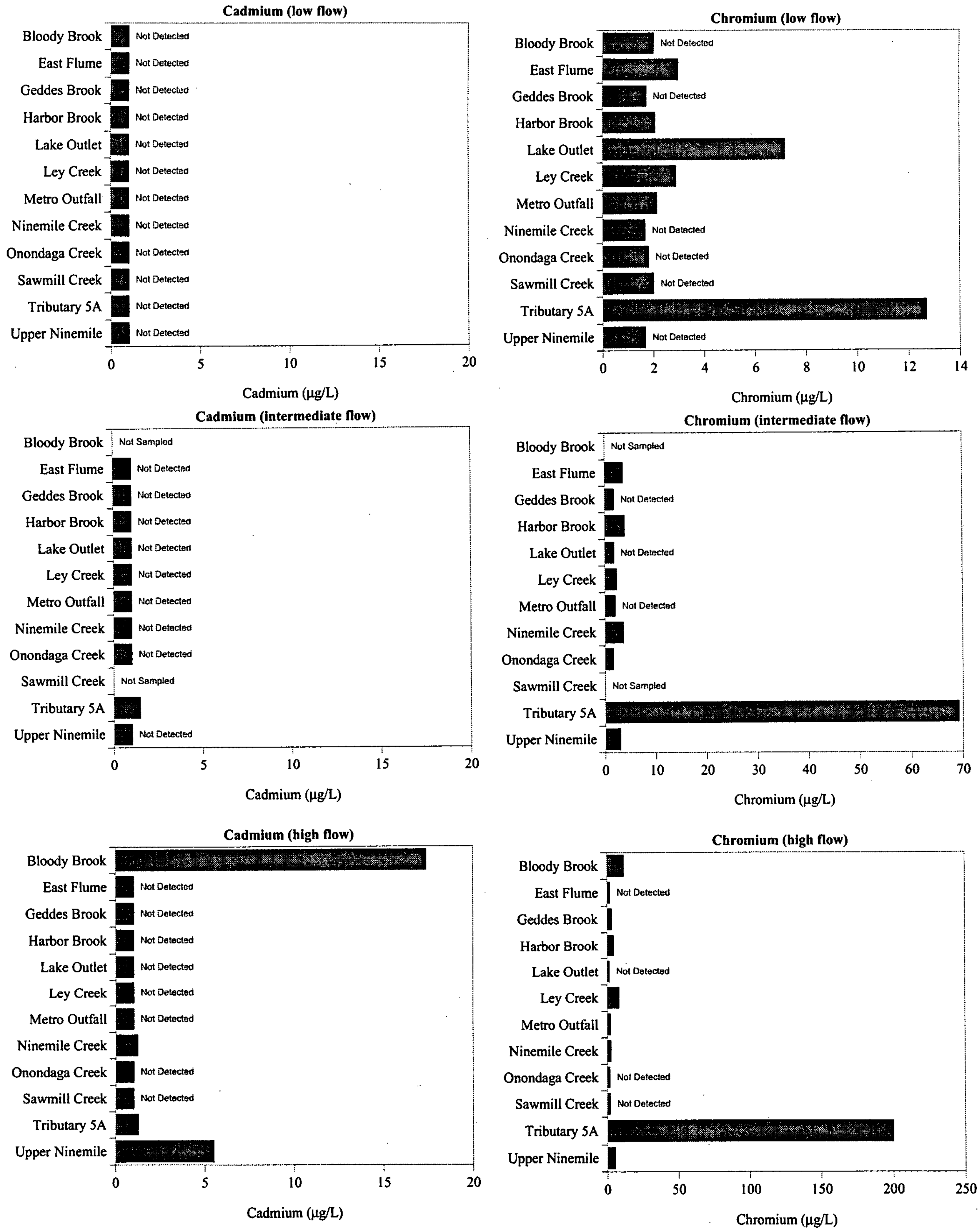
Figure 8-3
Total Mercury in Tributary Water and Metro Discharge During 1992



Notes:
1. Half the detection limits were used for non-detects.

TAMS

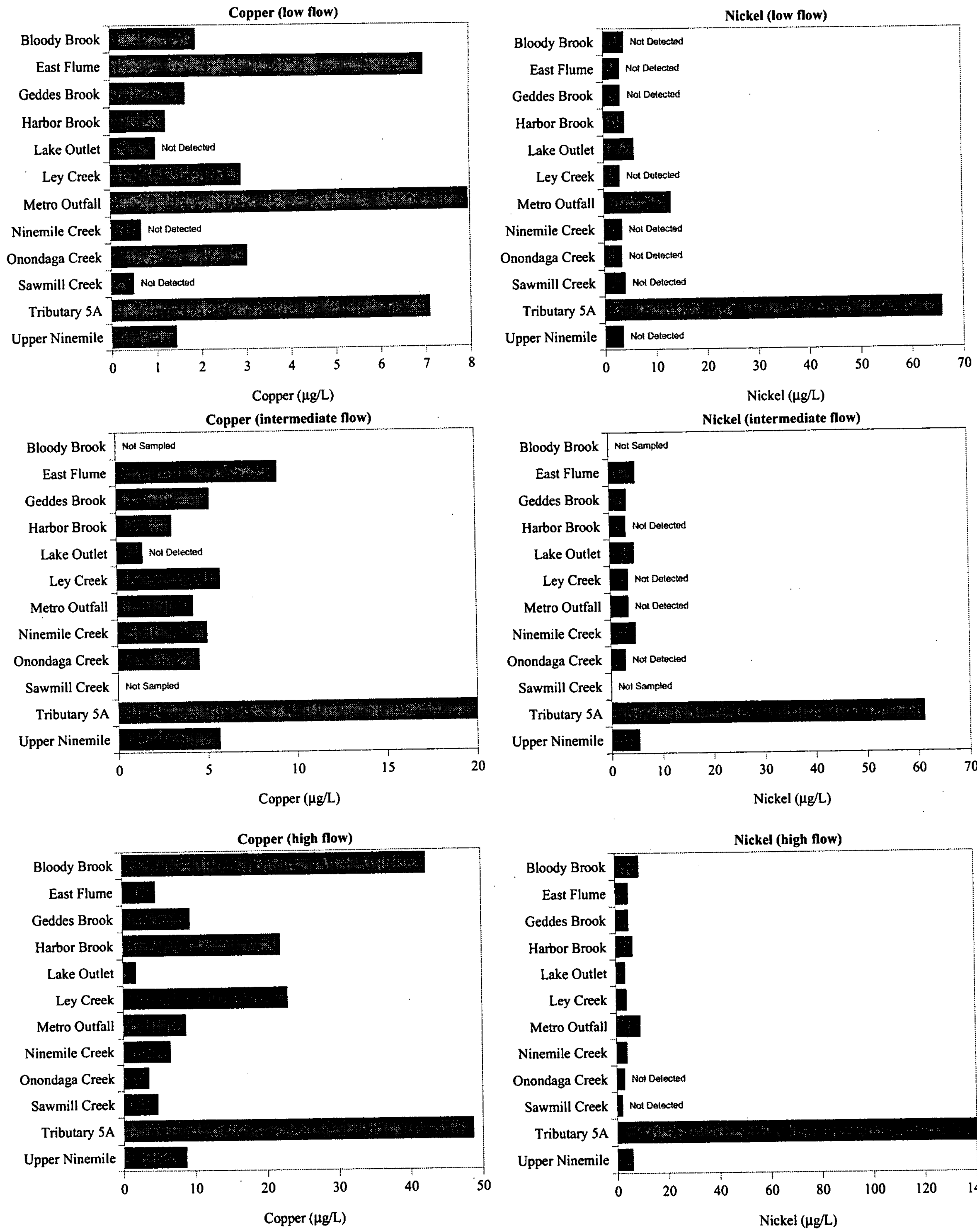
Figure 8-4
Total Methylmercury in Tributary Water and Metro Discharge During 1992



Notes:
1. Half the detection limits were used for non-detects.

TAMS

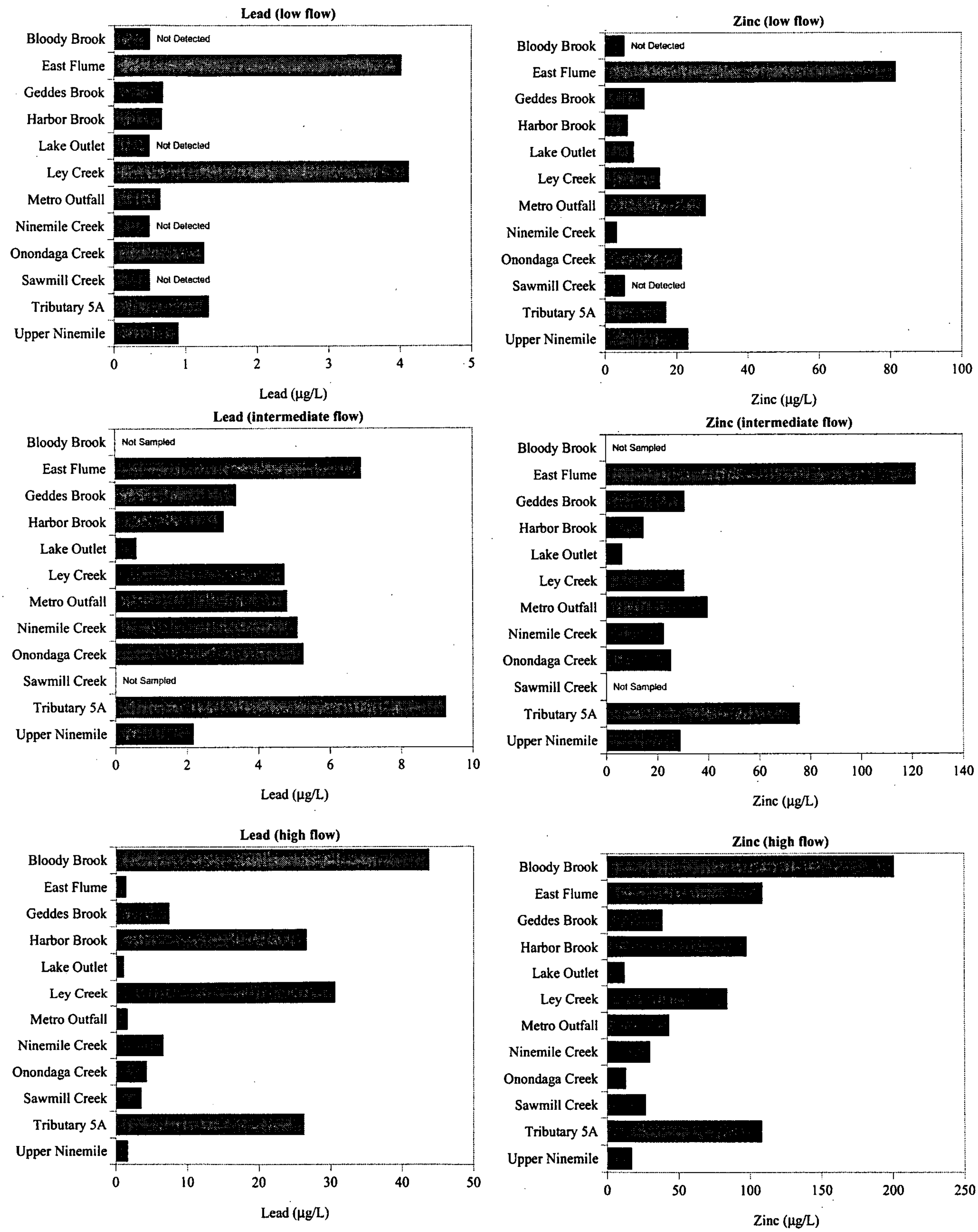
Figure 8-5
Mean Concentrations of Cadmium and Chromium in Tributary Water and Metro Discharge During 1992



Notes:
1. Half the detection limits were used for non-detects.

TAMS

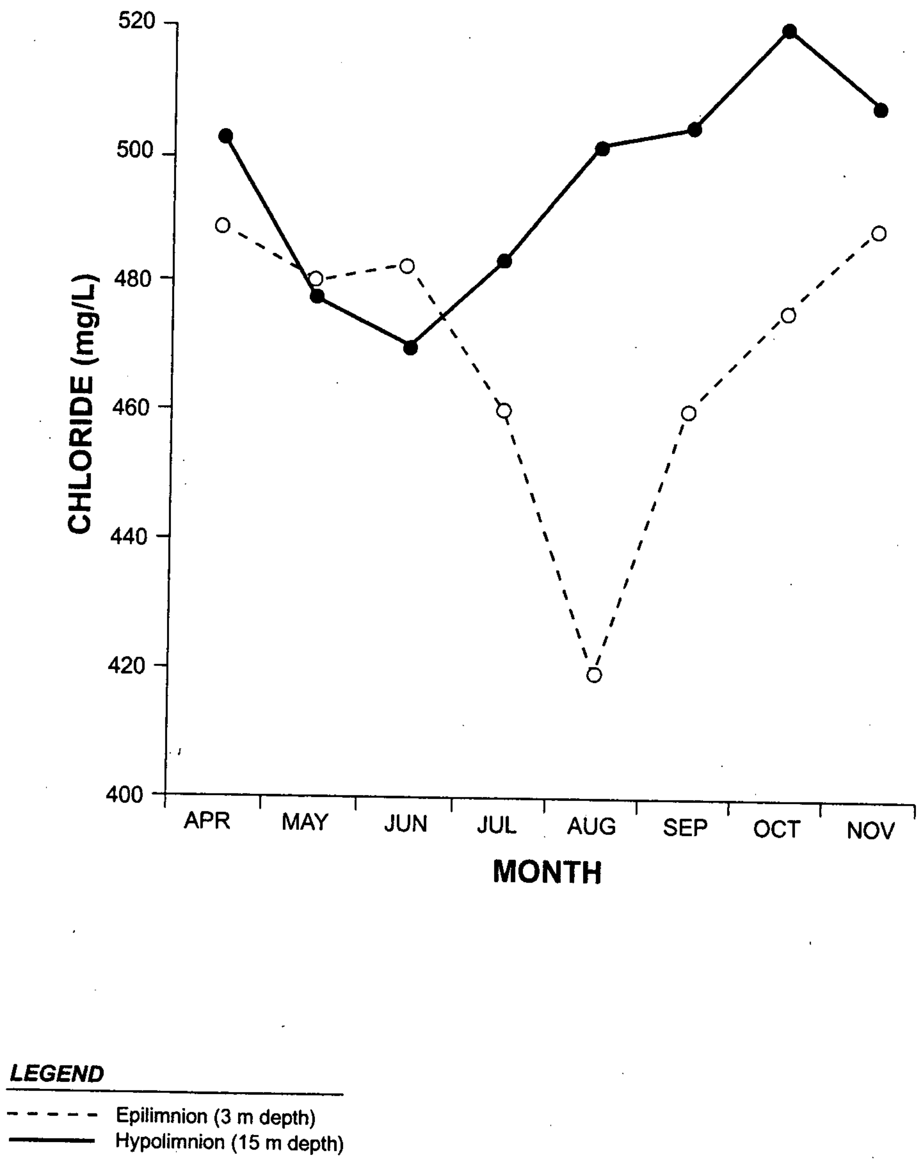
Figure 8-6
Mean Concentrations of Copper and Nickel in Tributary Water and Metro Discharge During 1992



Notes:
1. Half the detection limits were used for non-detects.

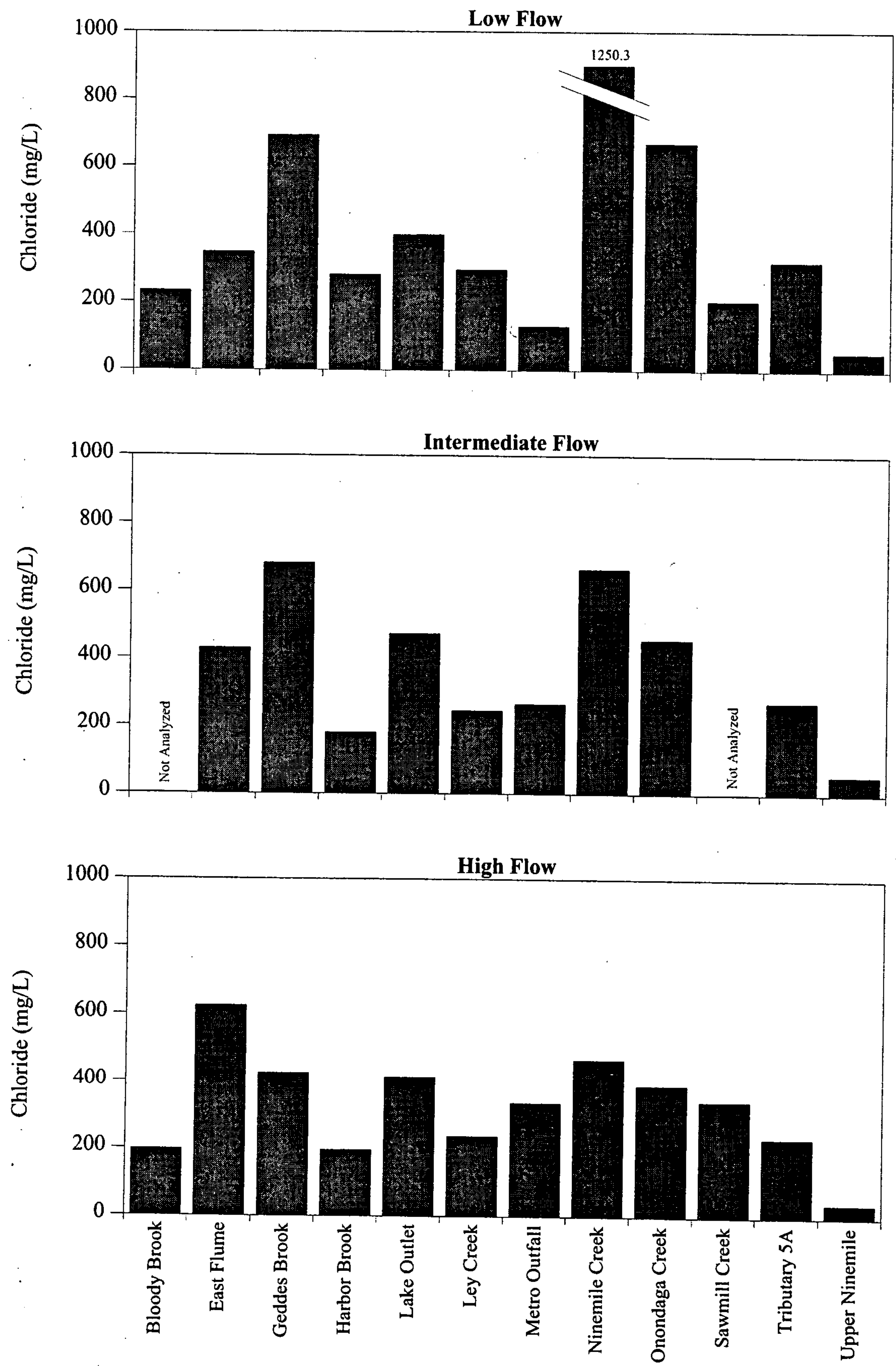
TAMS

Figure 8-7
Mean Concentrations of Lead and Zinc
in Tributary Water and Metro Discharge During 1992



Source: Stearns & Wheler (1994)
Exponent, 2001b

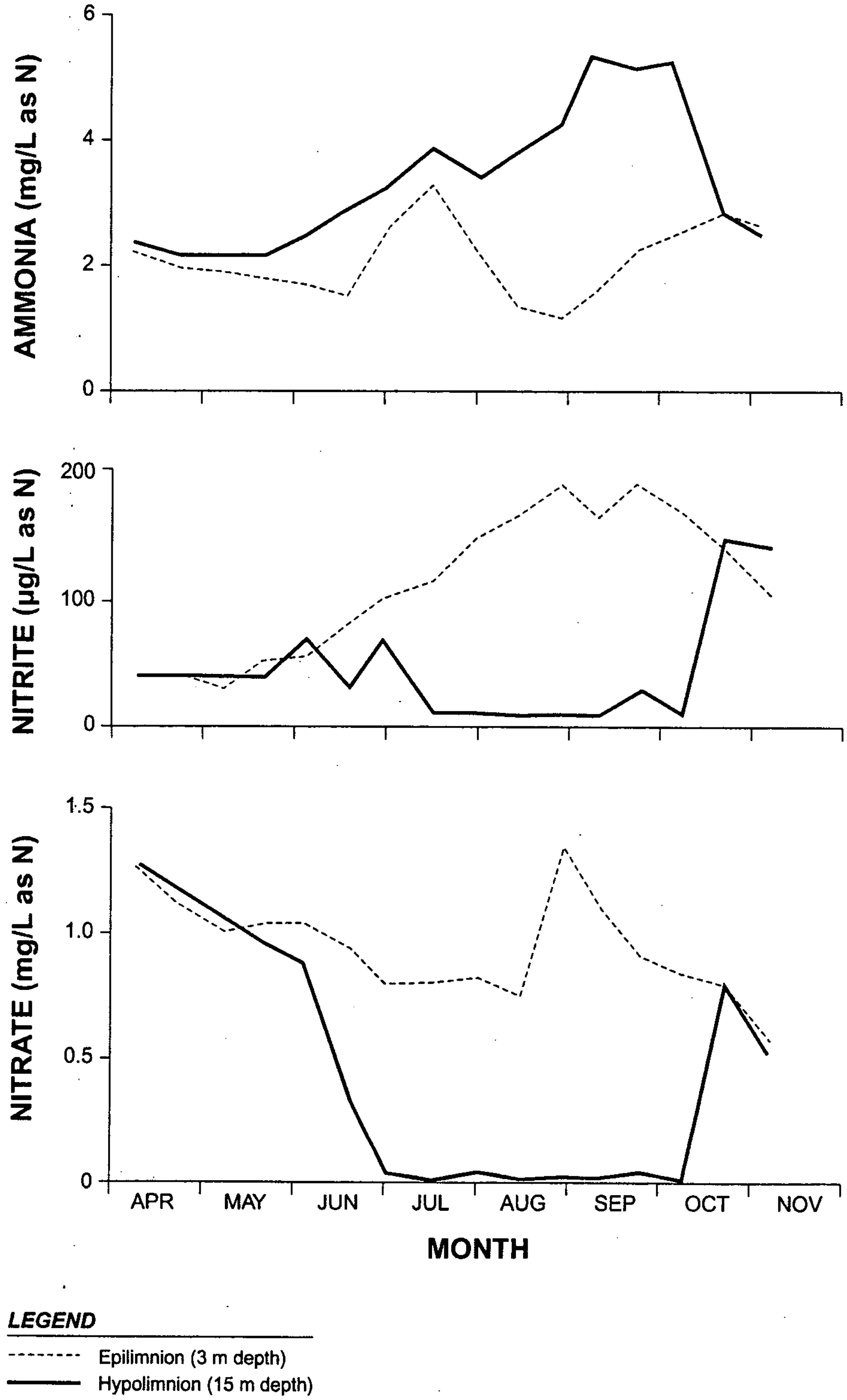
Figure 8-8. Concentrations of chloride in water of Onondaga Lake in 1992



Notes:
1. Half the detection limits were used for non-detects.

TAMS

Figure 8-9
Mean Concentrations of Chloride
in Tributary Water and Metro Discharge During 1992

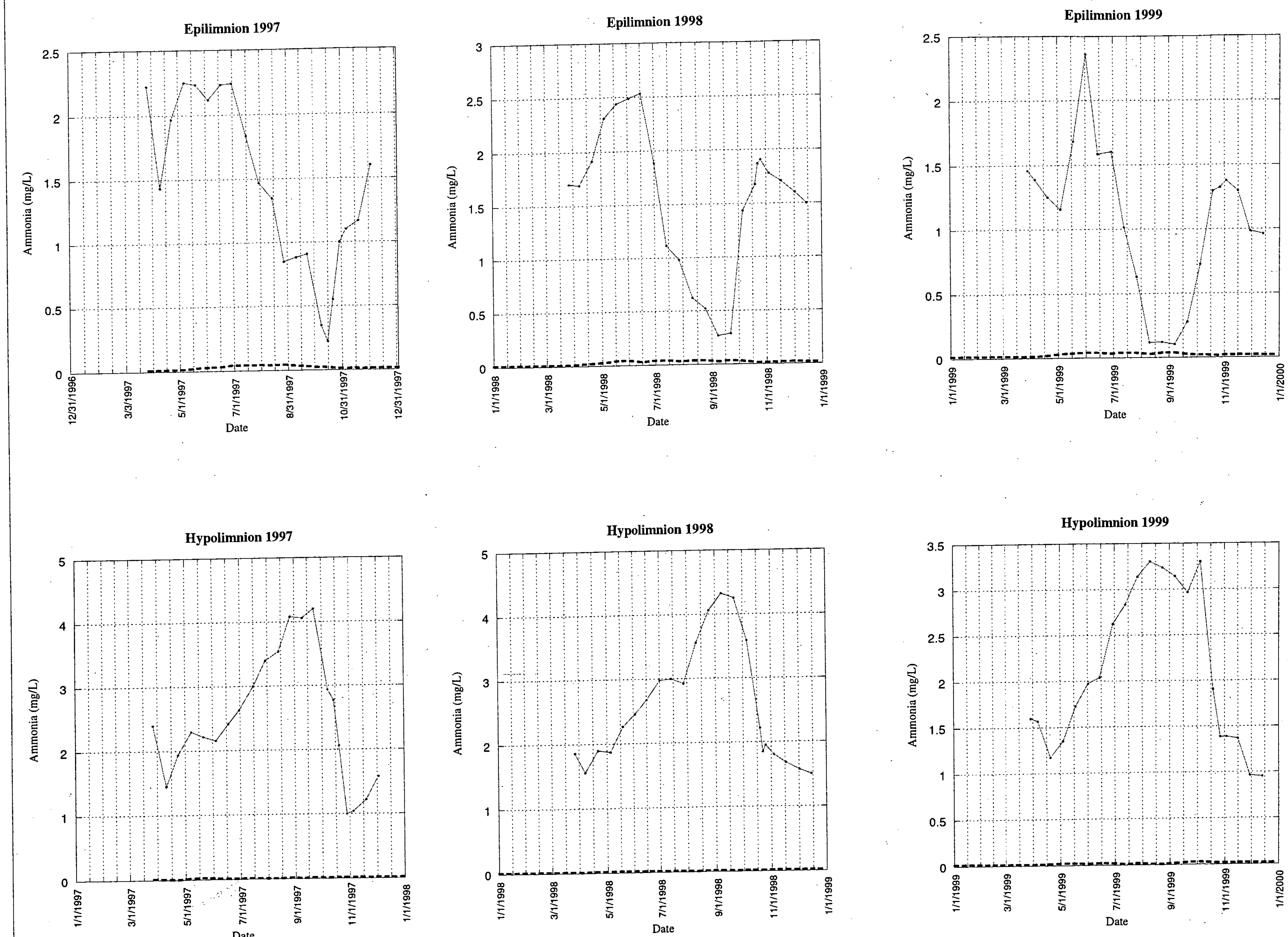


Source: Stearns & Wheler (1994)
Exponent, 2001b

Figure 8-10. Concentrations of ammonia, nitrite, and nitrate in Onondaga Lake in 1992

Figure 8-11

Ammonia Concentrations in Onondaga Lake 1997 - 2001



Note: New York State chronic water quality standard is a function of pH and water temperature (NYSDEC, 1998)

Points shown are averages for all epilimnion or hypolimnion samples available on each sample date.

Source: Onondaga County Ambient Water Quality Monitoring Program.

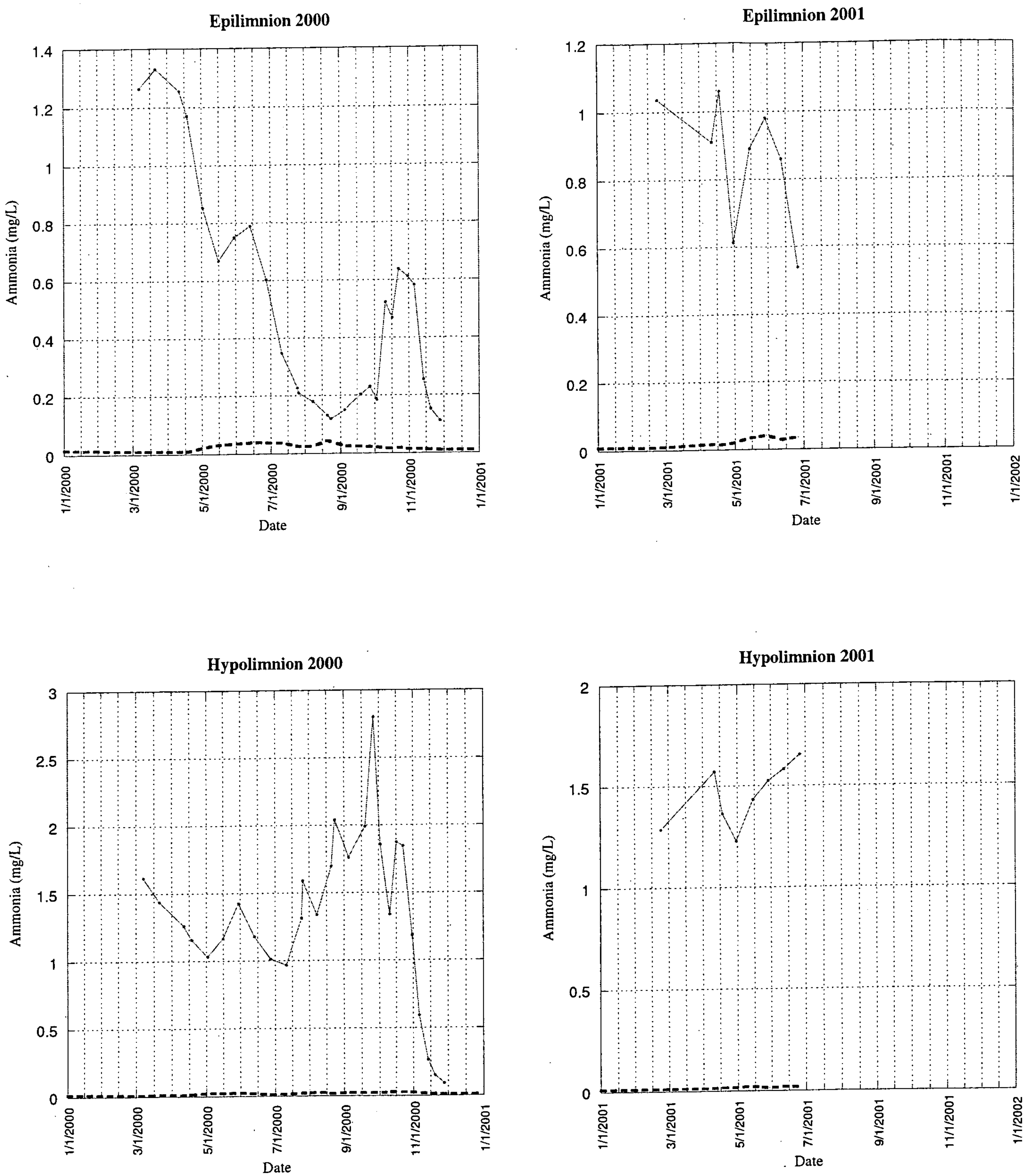


Figure 8-11 cont.

Ammonia Concentrations in Onondaga Lake 1997 - 2001

Ammonia
NYS Standard

Note: New York State chronic water quality standard is a function of pH and water temperature (NYSDEC, 1998)

Points shown are averages for all epilimnion or hypolimnion samples available on each sample date.

Source: Onondaga County Ambient Water Quality Monitoring Program.

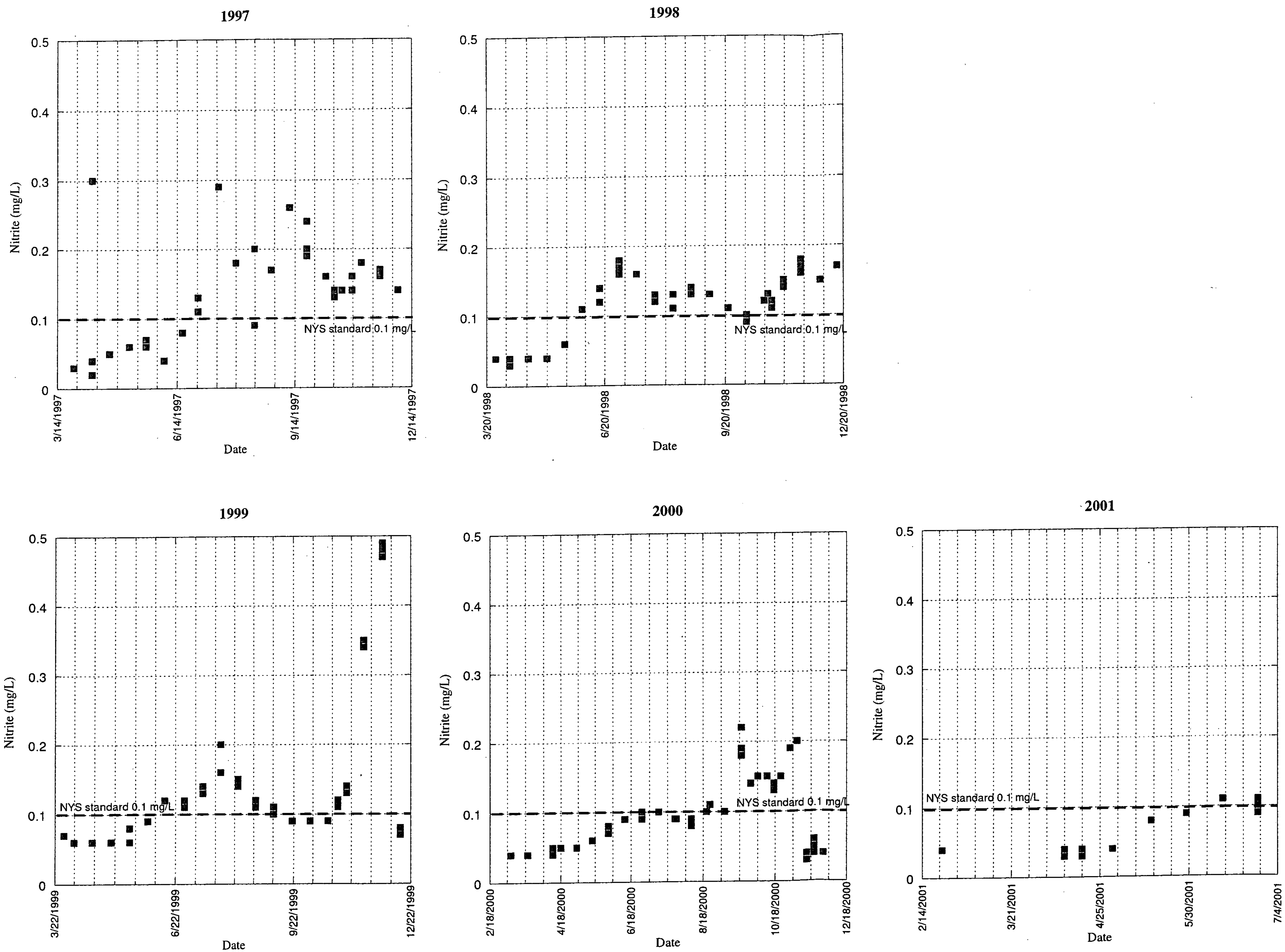
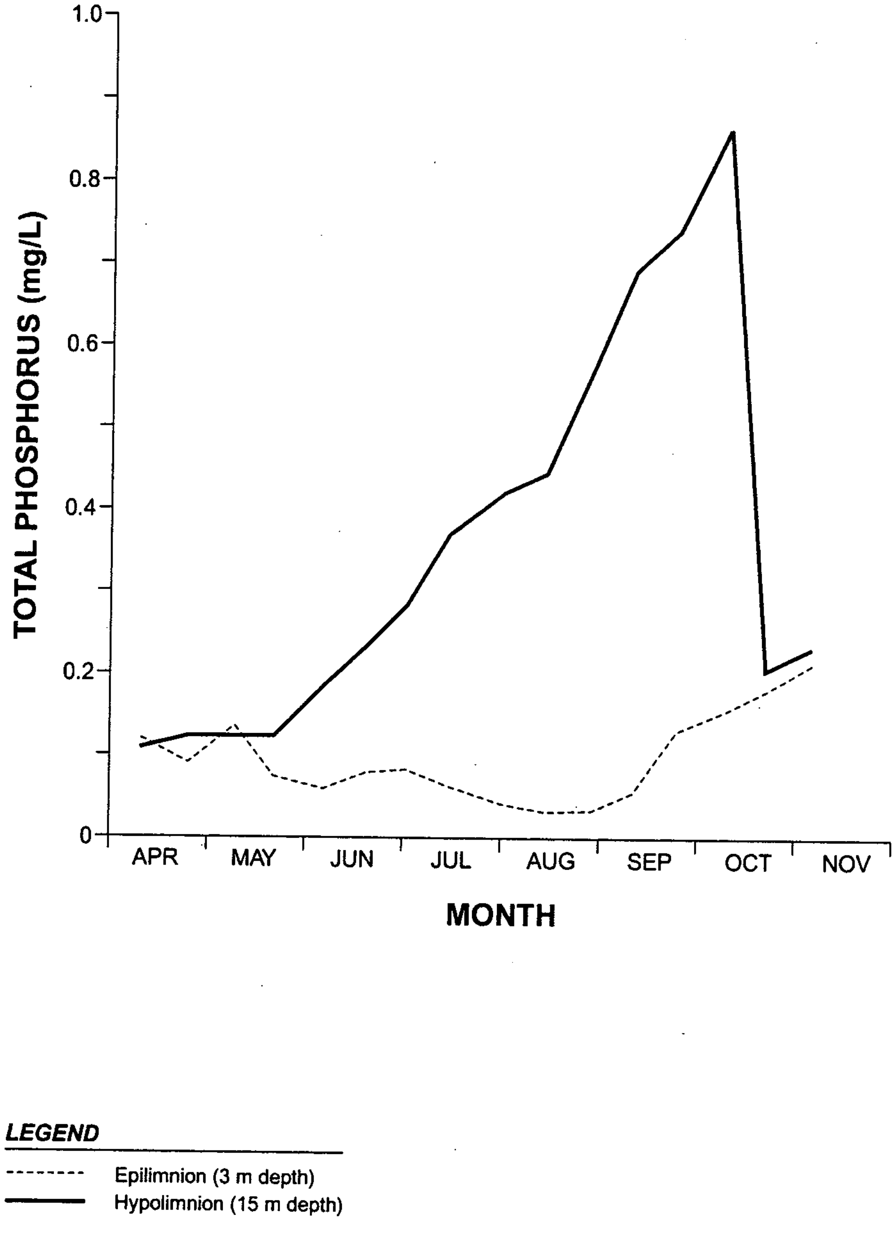


Figure 8-12

Nitrite Concentrations in the Epilimnion (above 9 m depths) of Onondaga Lake, 1997 - 2001

Note: Data shown can be from several sample depths. Where this occurs, the points are shown independently of each other on the plot.

Source: Onondaga County Ambient Water Quality Monitoring Program



Source: Stearns & Wheler (1994)
Exponent, 2001b

Figure 8-13. Concentrations of total phosphorus in Onondaga Lake in 1992

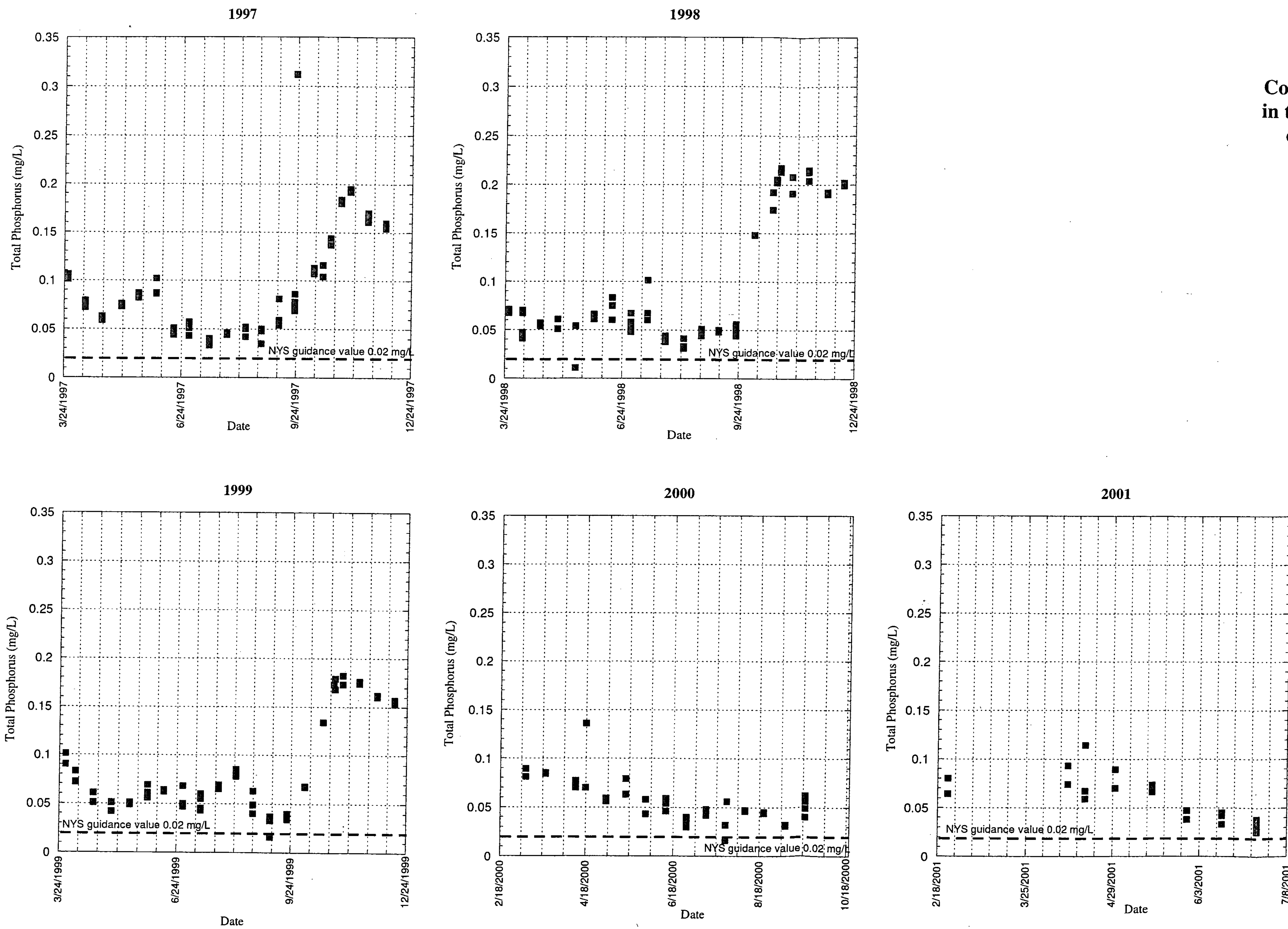
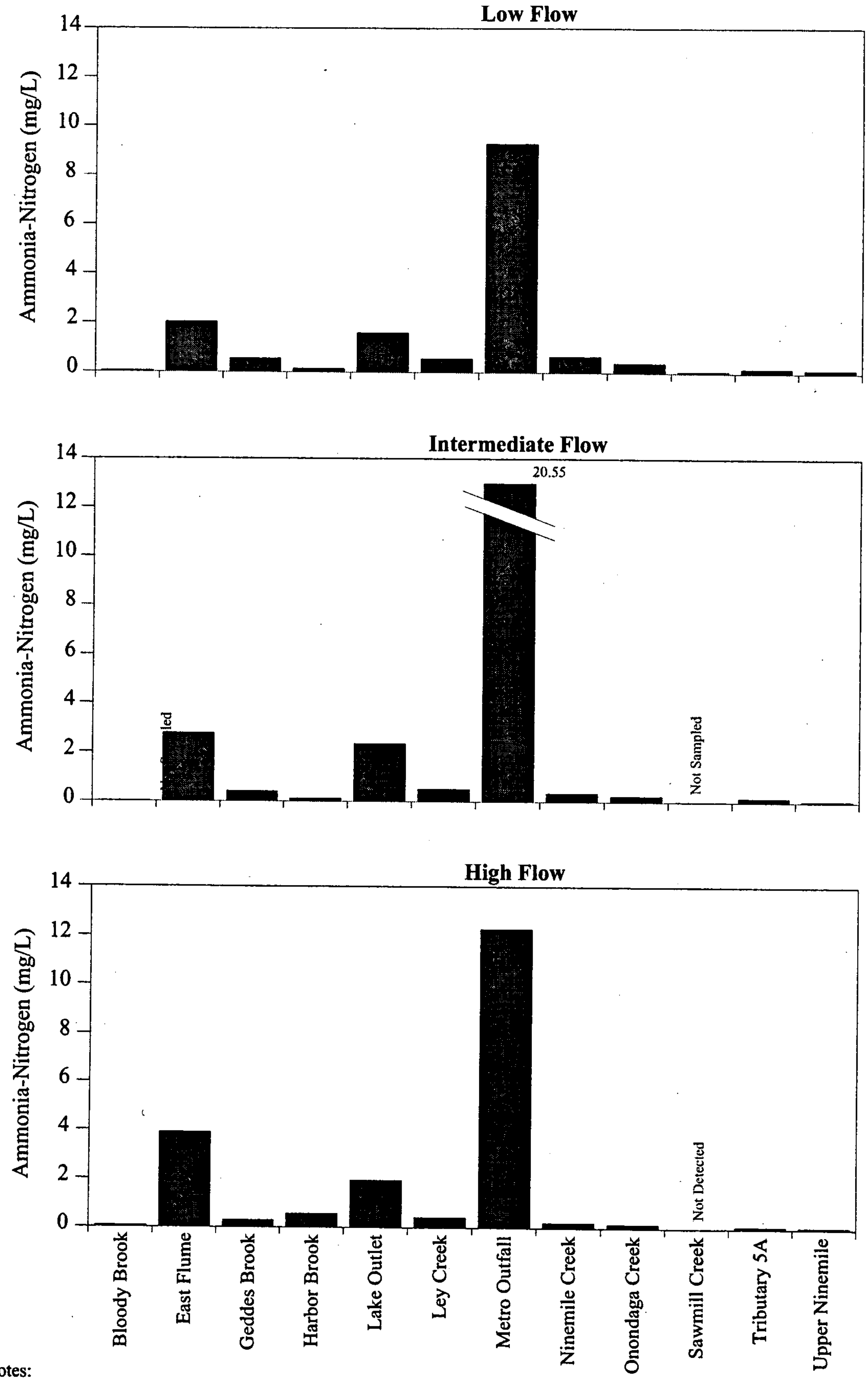


Figure 8-14

Concentrations of Total Phosphorus in the Epilimnion (above 9 m depth) of Onondaga Lake, 1997 - 2001

Note: Data shown can be from several sample depths. Where this occurs, the points are shown independently of each other on the plot.

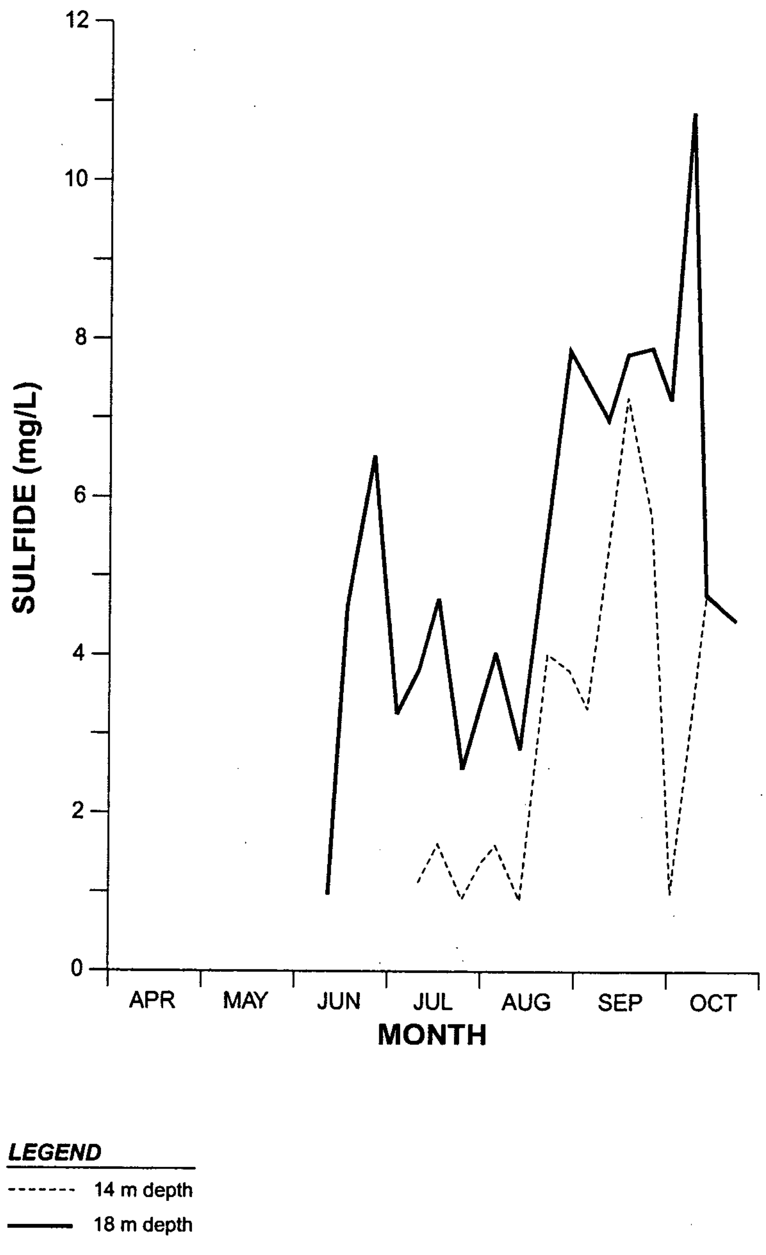
Source: Onondaga County Ambient Water Quality Monitoring Data



Notes:
1. Half the detection limits were used for non-detects.

TAMS

Figure 8-15
Mean Concentrations of Ammonia-Nitrogen in Tributary Water and Metro Discharge During 1992



Source: Stearns & Wheler (1994)
Exponent, 2001b

Figure 8-16. Concentrations of sulfide in Onondaga Lake in 1992

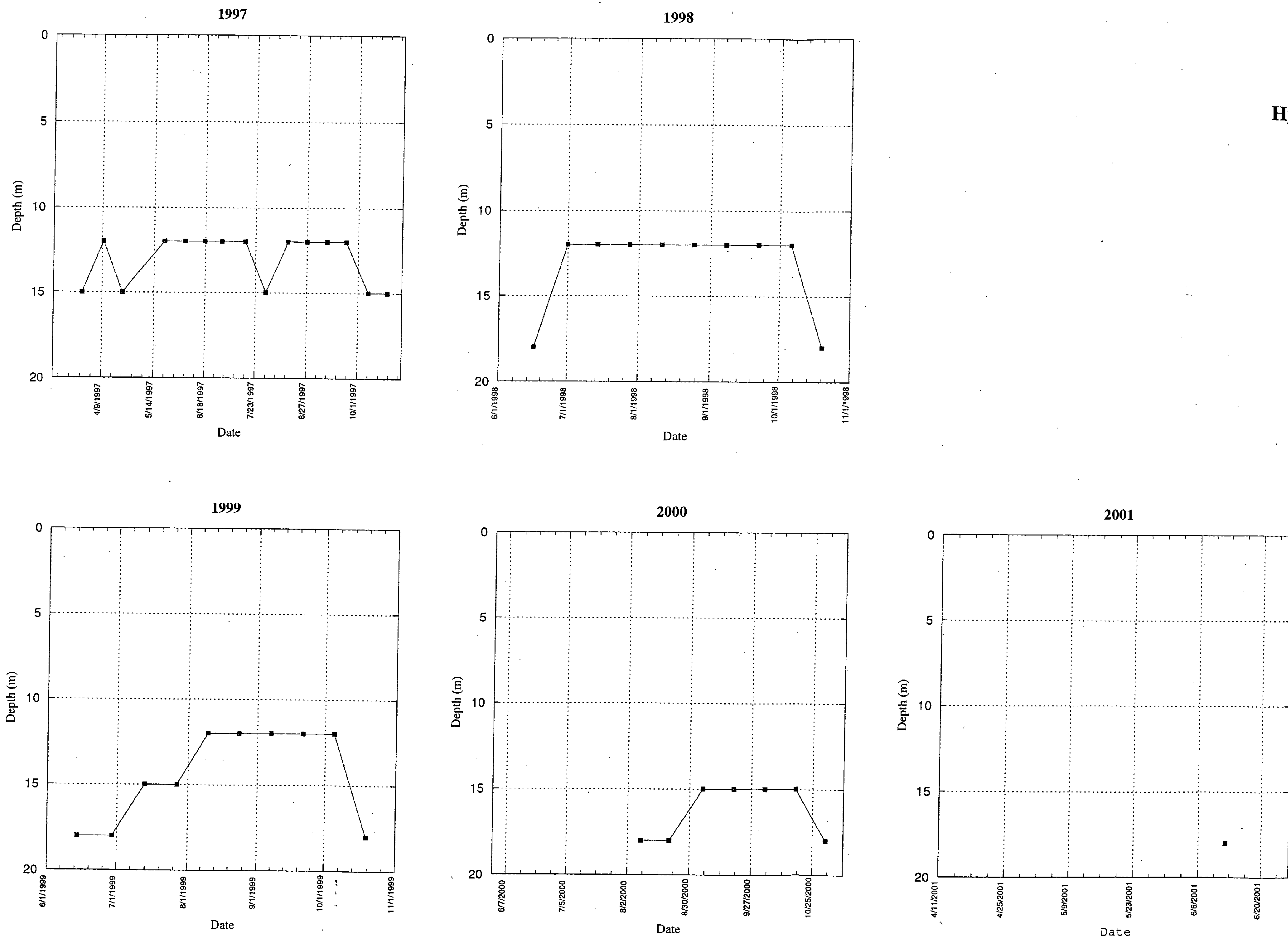
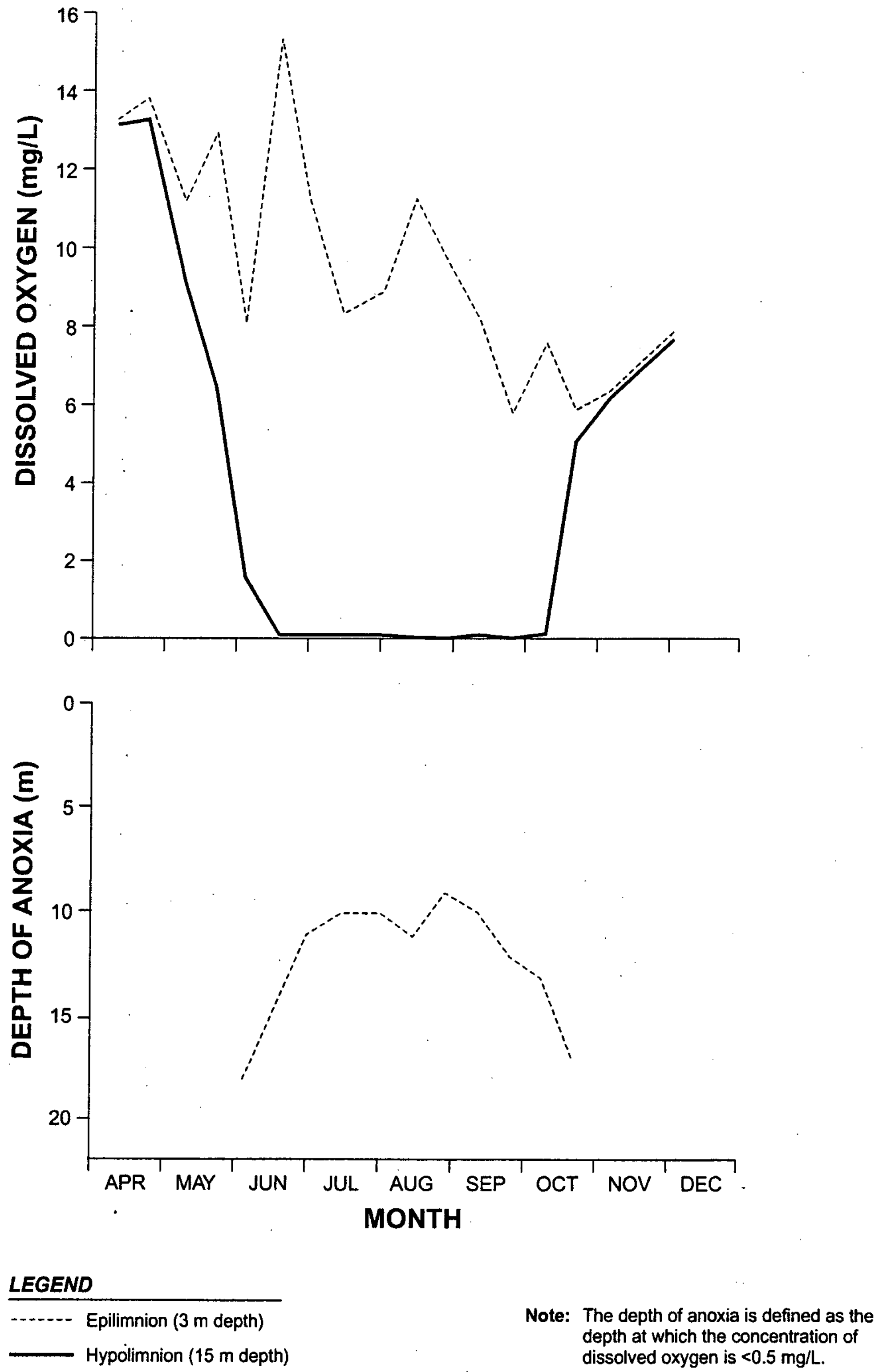


Figure 8-17

Hypolimnion Sulfide Concentrations in Onondaga Lake, 1997 - 2001

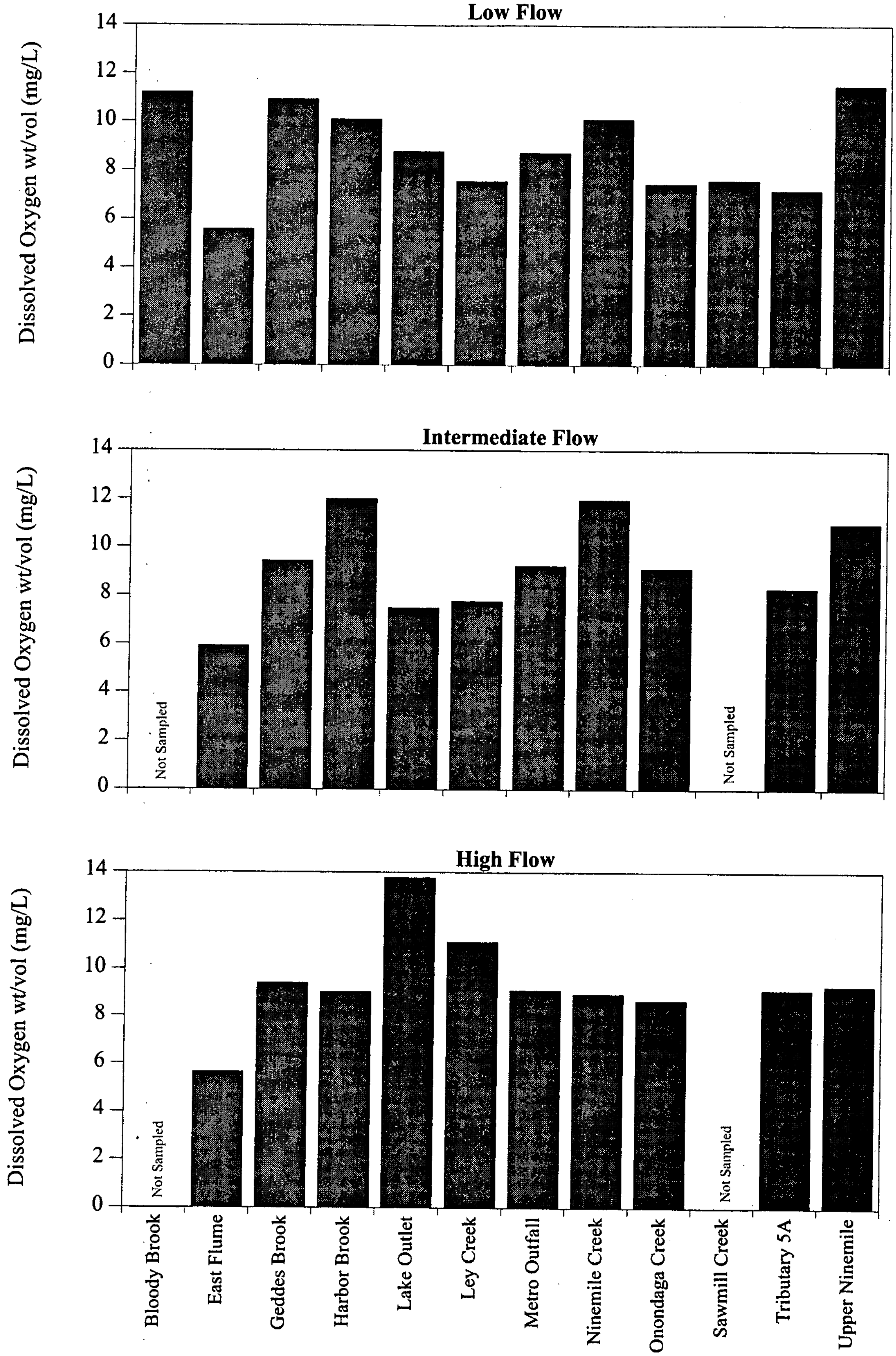
Note: Data shown are the uppermost depth at which the standard was exceeded.

Source: Onondaga County Ambient Water Quality Monitoring Data



Source: Stearns & Wheler (1994)
Exponent, 2001b

Figure 8-18. Concentrations of dissolved oxygen and depths of anoxia in Onondaga Lake in 1992



Notes:
1. Half the detection limits were used for non-detects.

TAMS

Figure 8-19
Mean Concentrations of Dissolved Oxygen in Tributary Water and Metro Discharge During 1992

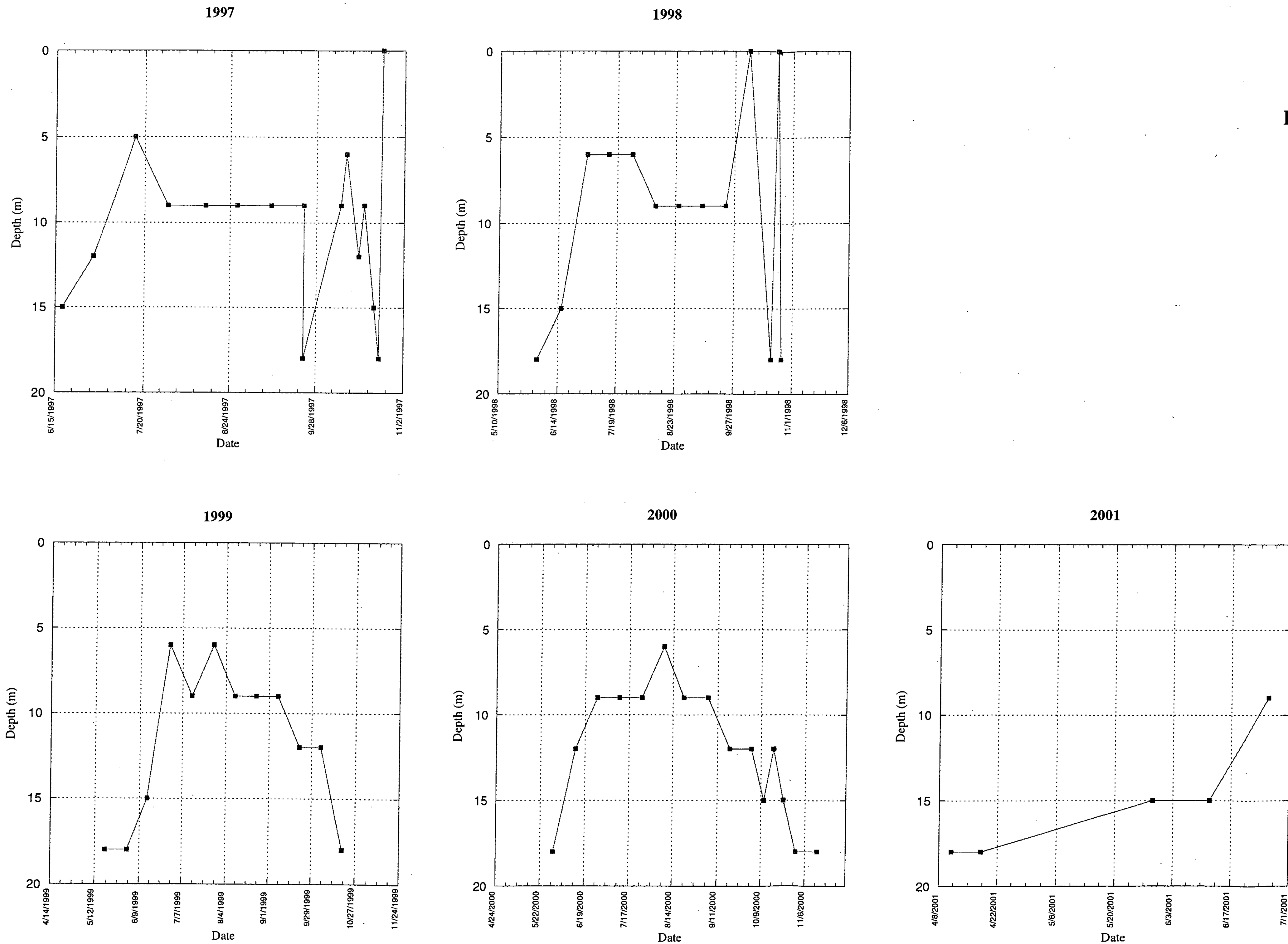
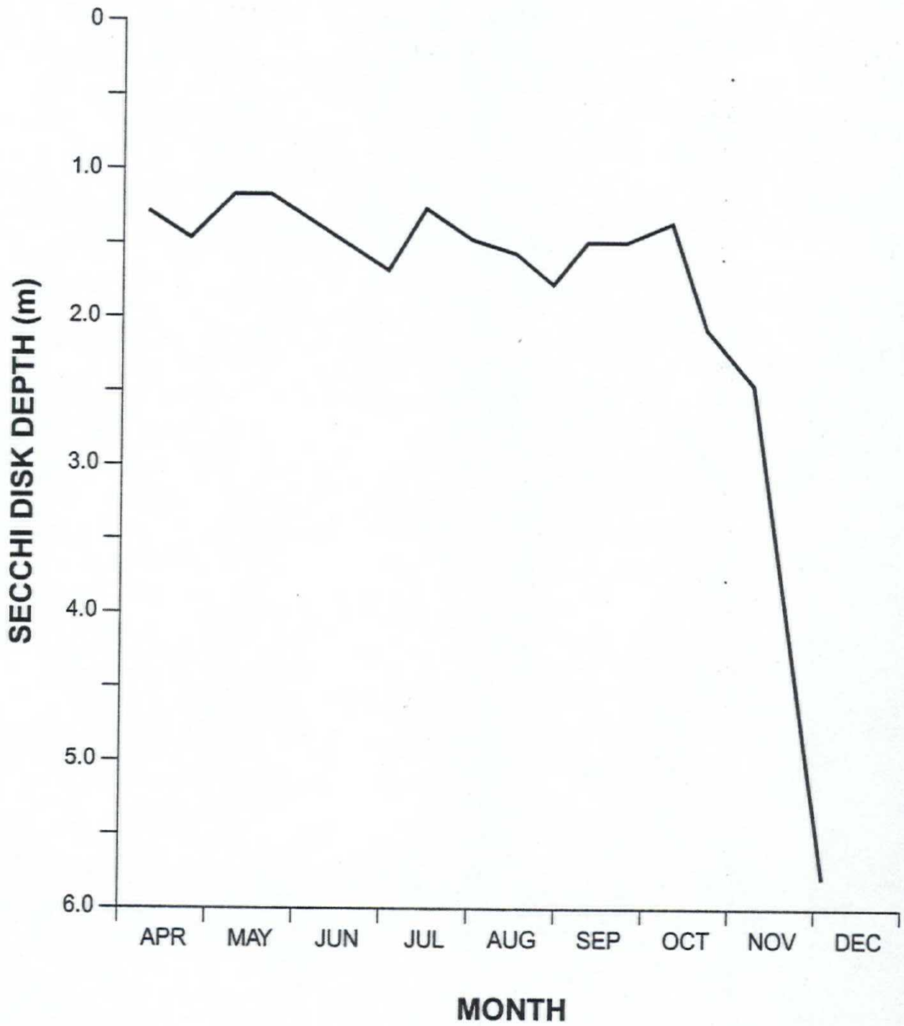


Figure 8-20

Dissolved Oxygen Concentrations in Onondaga Lake, 1997 - 2001

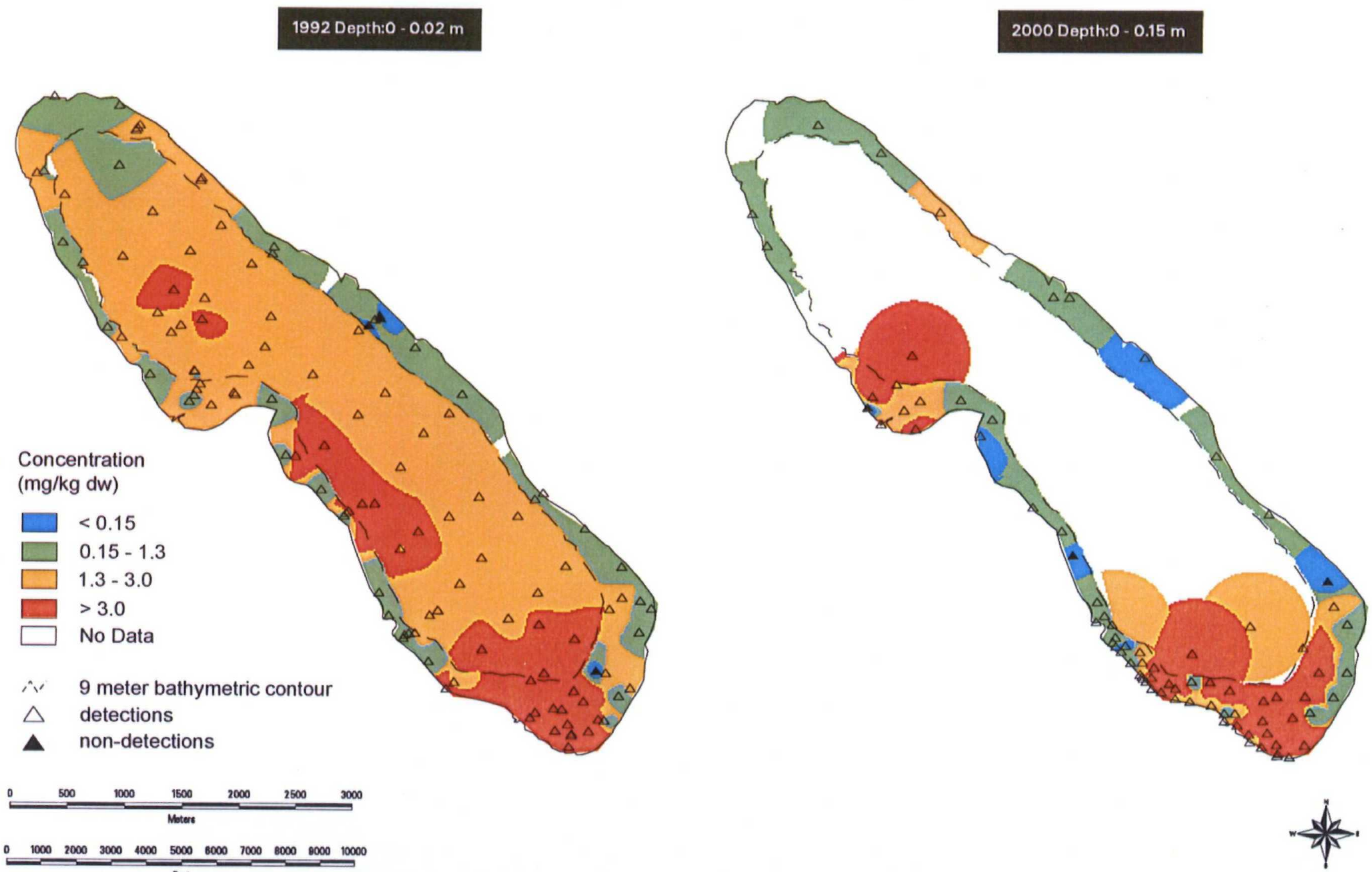
Note: Data shown are from uppermost depth at which the standard was exceeded.

Source: Onondaga County Ambient Water Quality Monitoring Data



Source: Stearns & Wheler (1994)
Exponent, 2001b

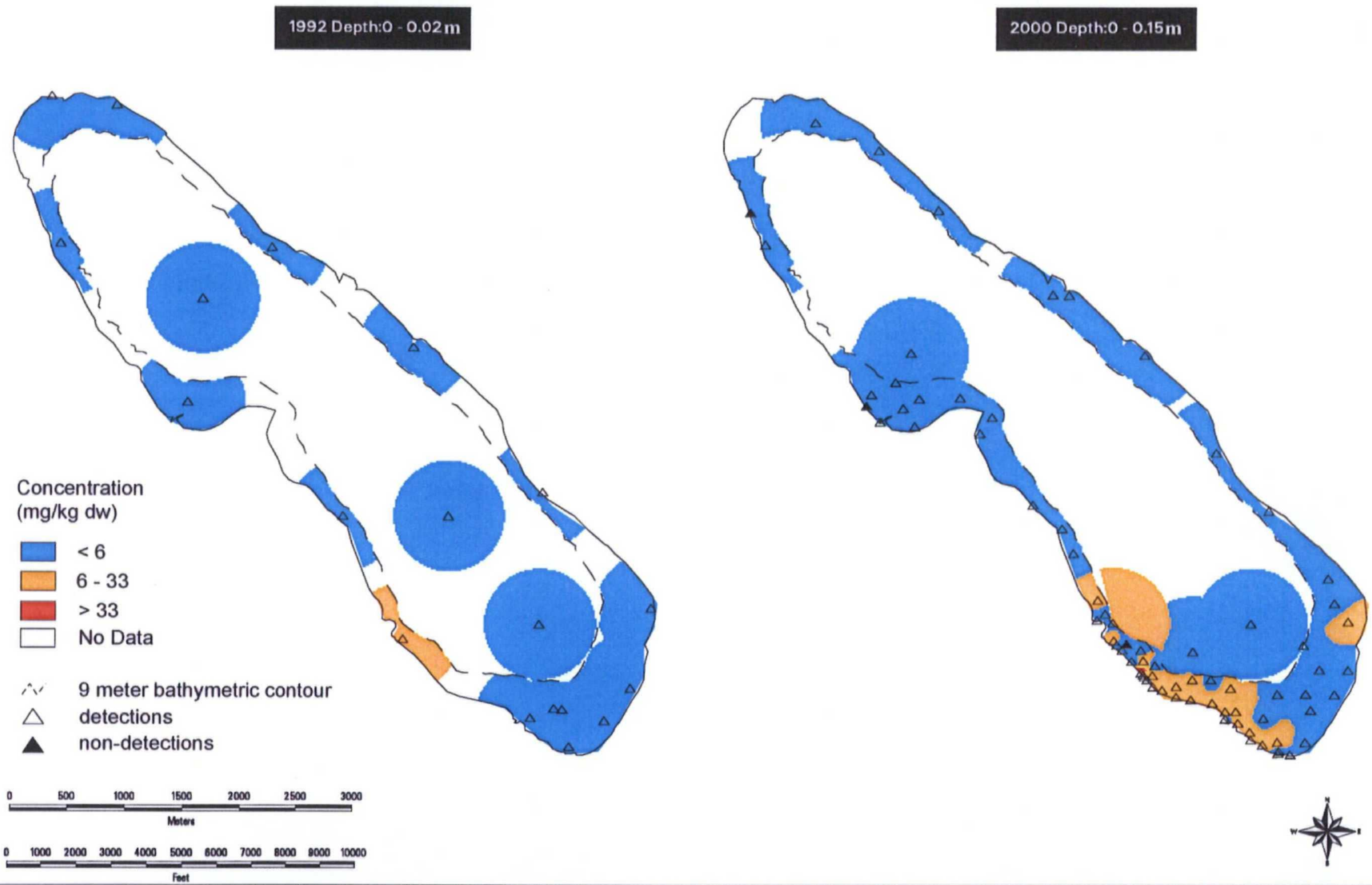
Figure 8-21. Secchi disk depths in Onondaga Lake in 1992



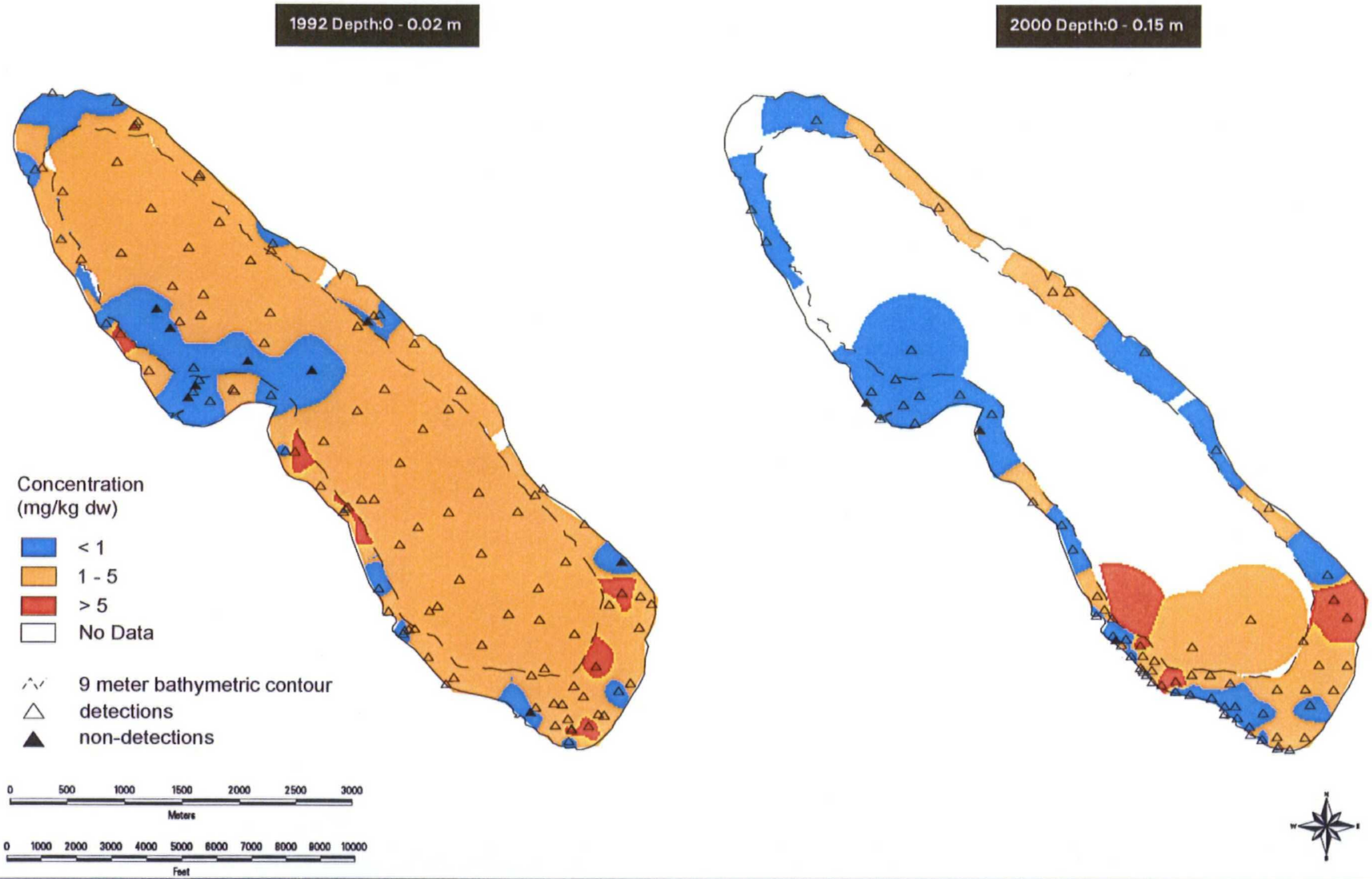
Onondaga Lake BERA

Figure 8-22. Distribution of Mercury in Surface Sediment of Onondaga Lake in 1992 and 2000

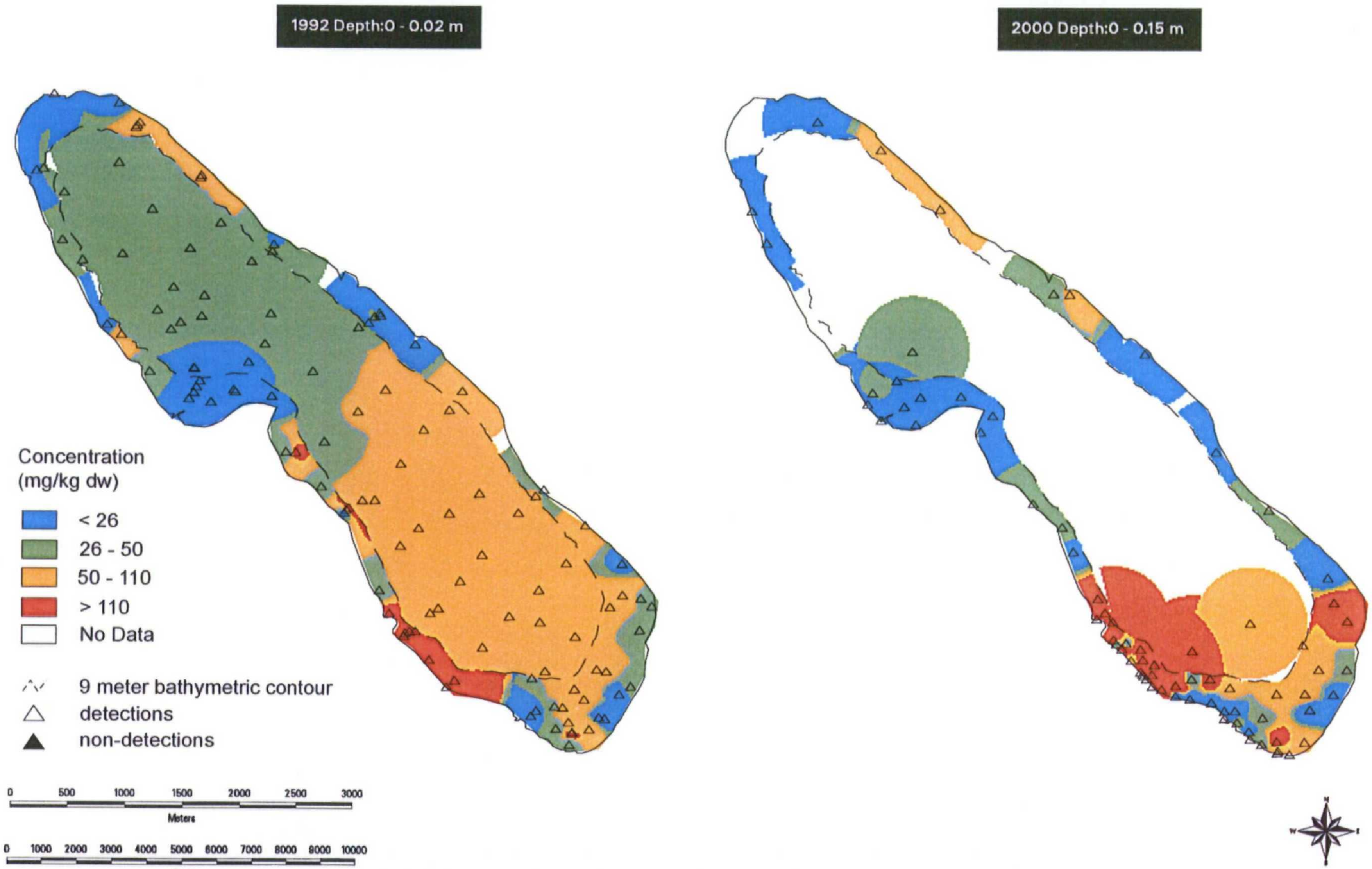
TAMS



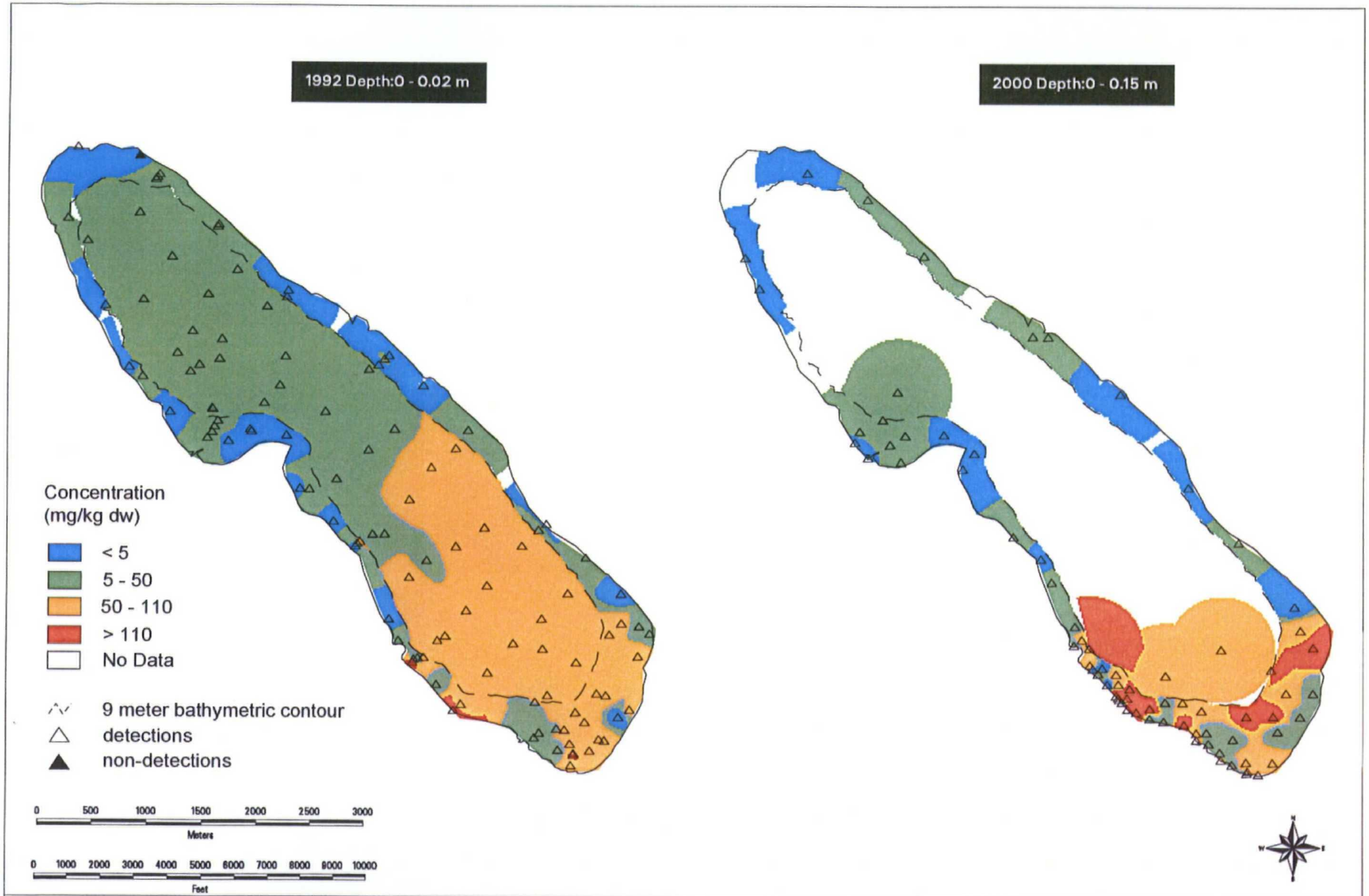
Onondaga Lake BERA	Figure 8-23. Distribution of Arsenic in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



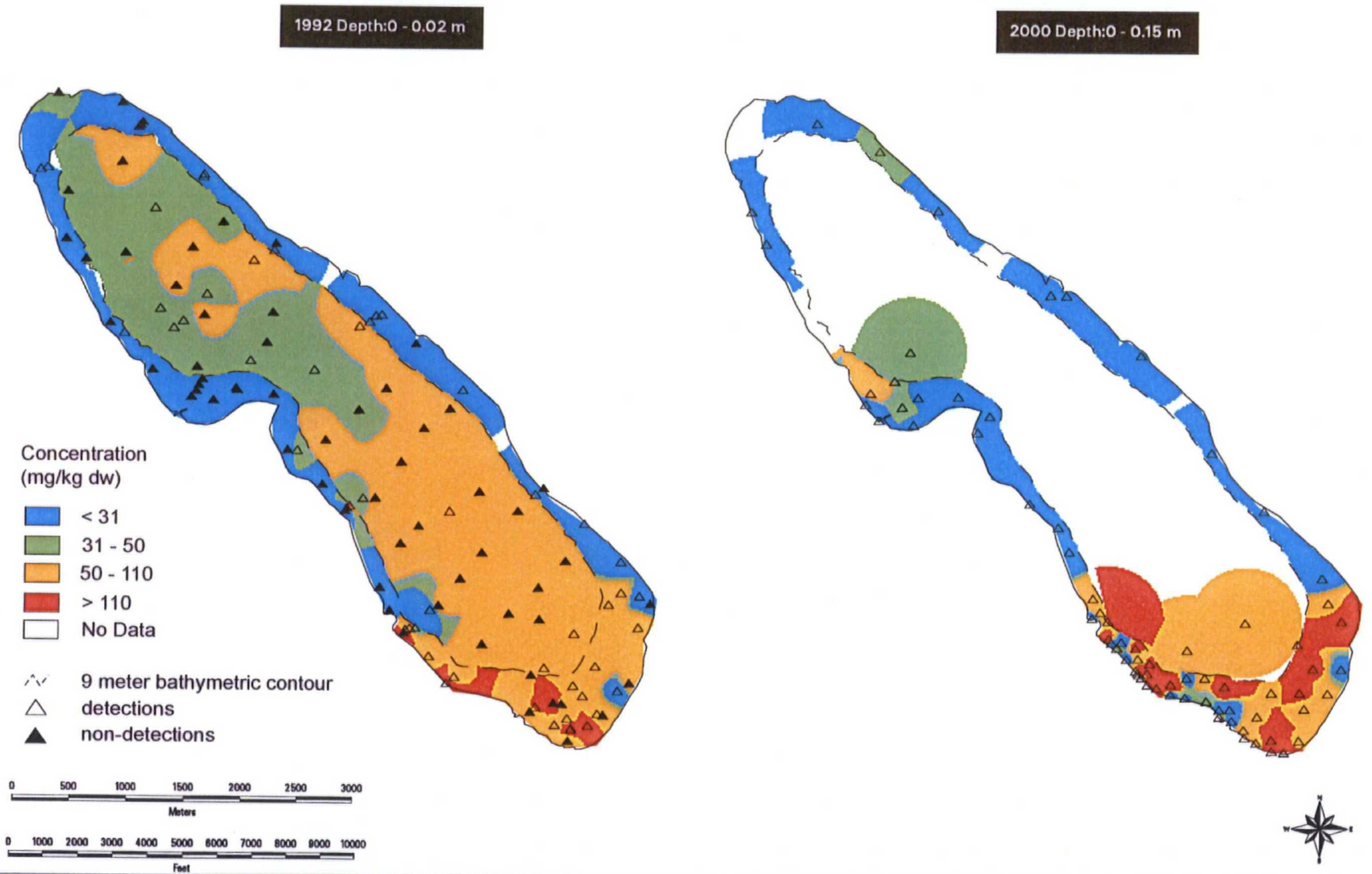
Onondaga Lake BERA	Figure 8-24. Distribution of Cadmium in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



Onondaga Lake BERA	Figure 8-25. Distribution of Chromium in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



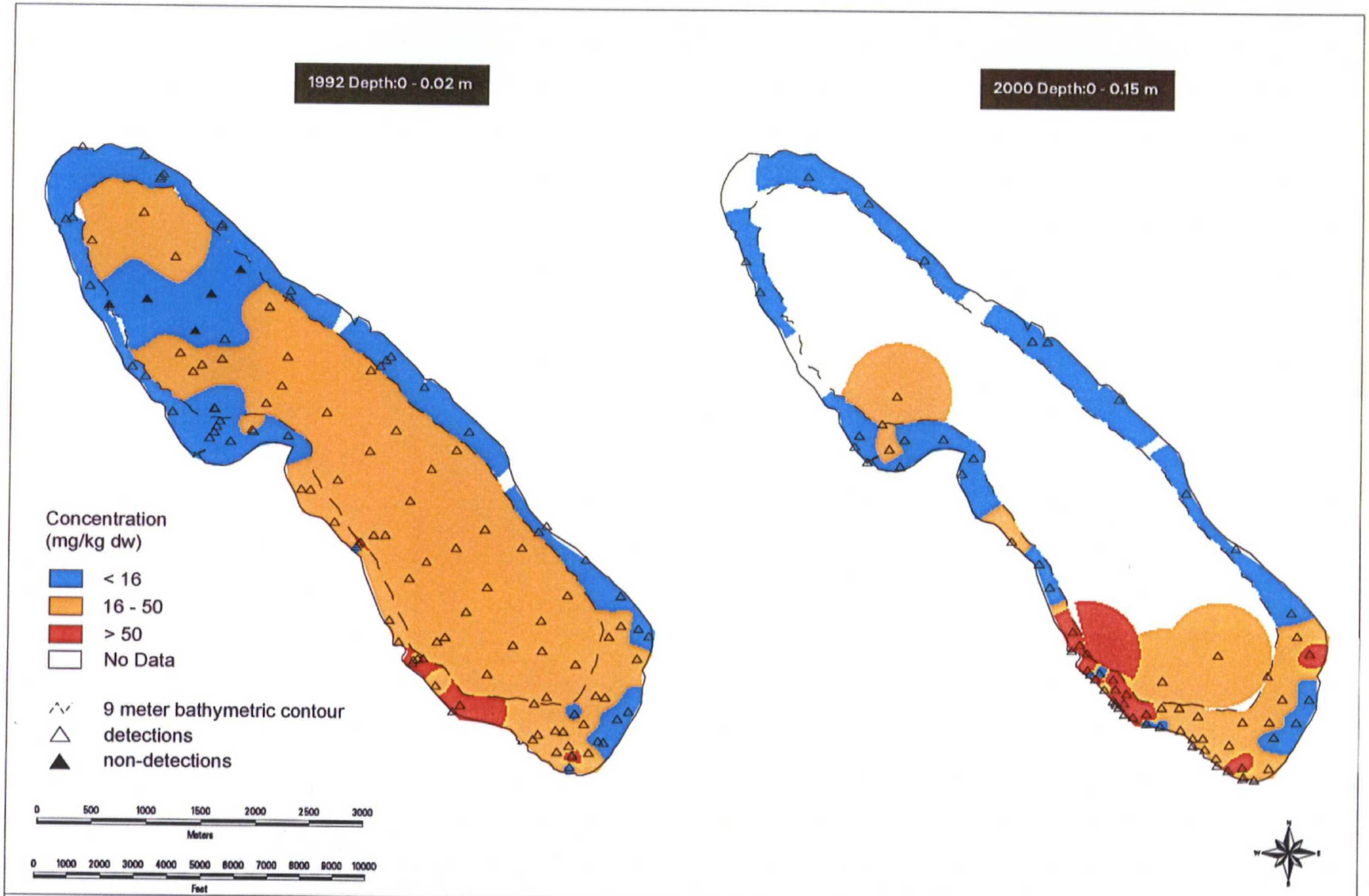
Onondaga Lake BERA	Figure 8-26. Distribution of Copper in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



Onondaga Lake BERA

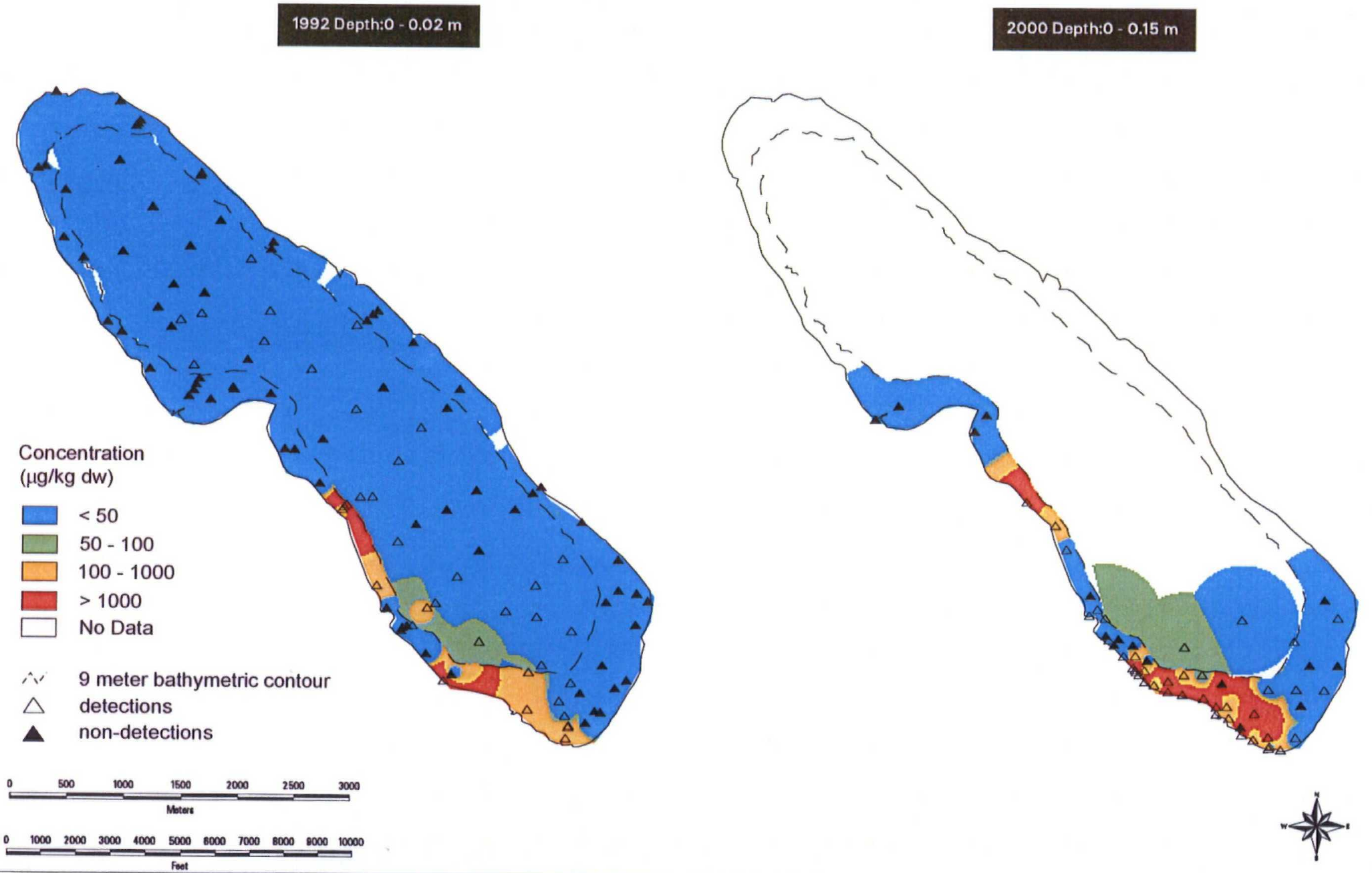
Figure 8-27. Distribution of Lead in Surface Sediments of Onondaga Lake in 1992 and 2000

TAMS



Onondaga Lake BERA	Figure 8-28. Distribution of Nickel in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

Onondaga Lake BERA	Figure 8-29. Distribution of Zinc in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

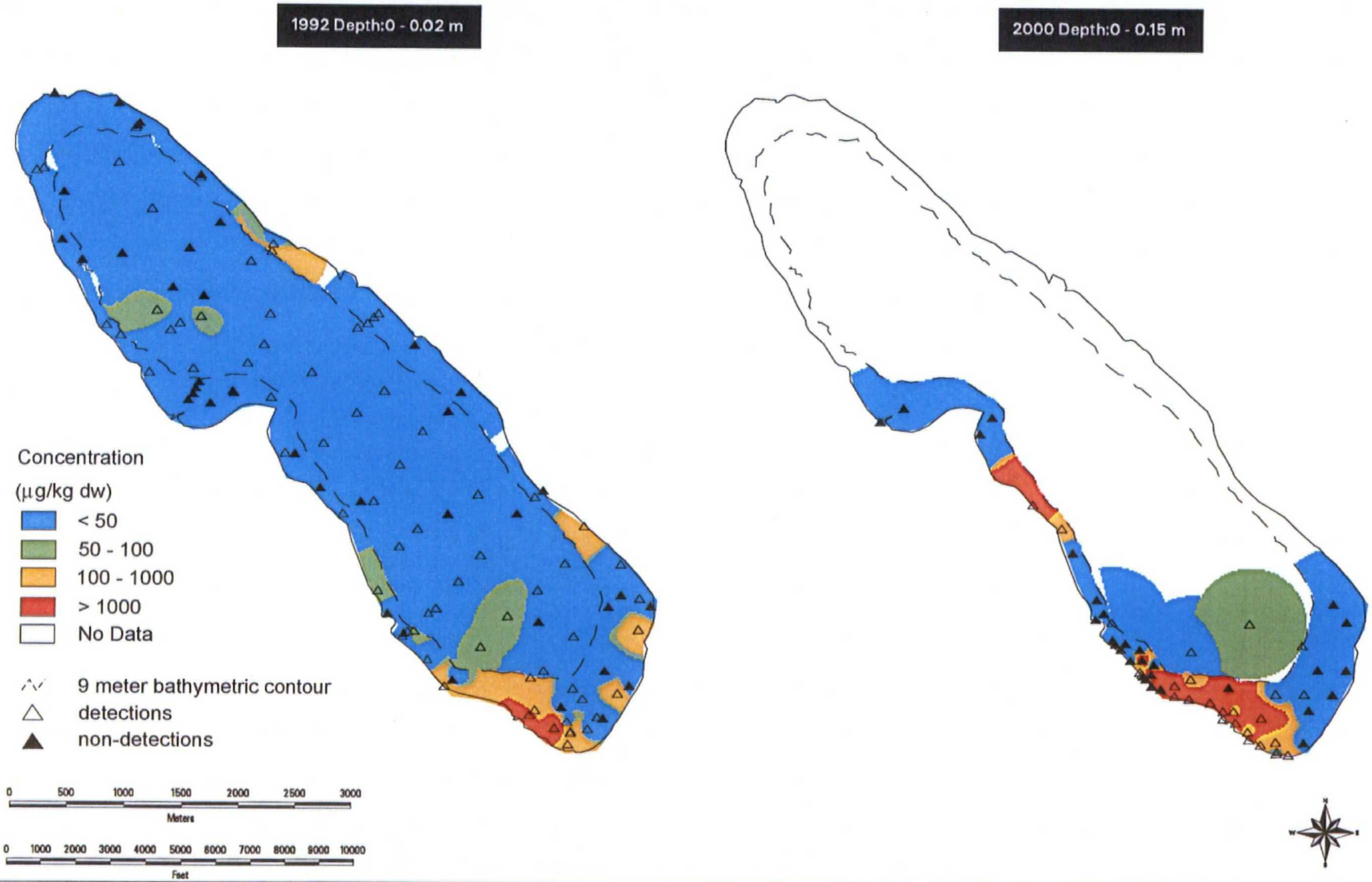


Onondaga Lake BERA	Figure 8-30. Distribution of Benzene in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

Onondaga Lake BERA

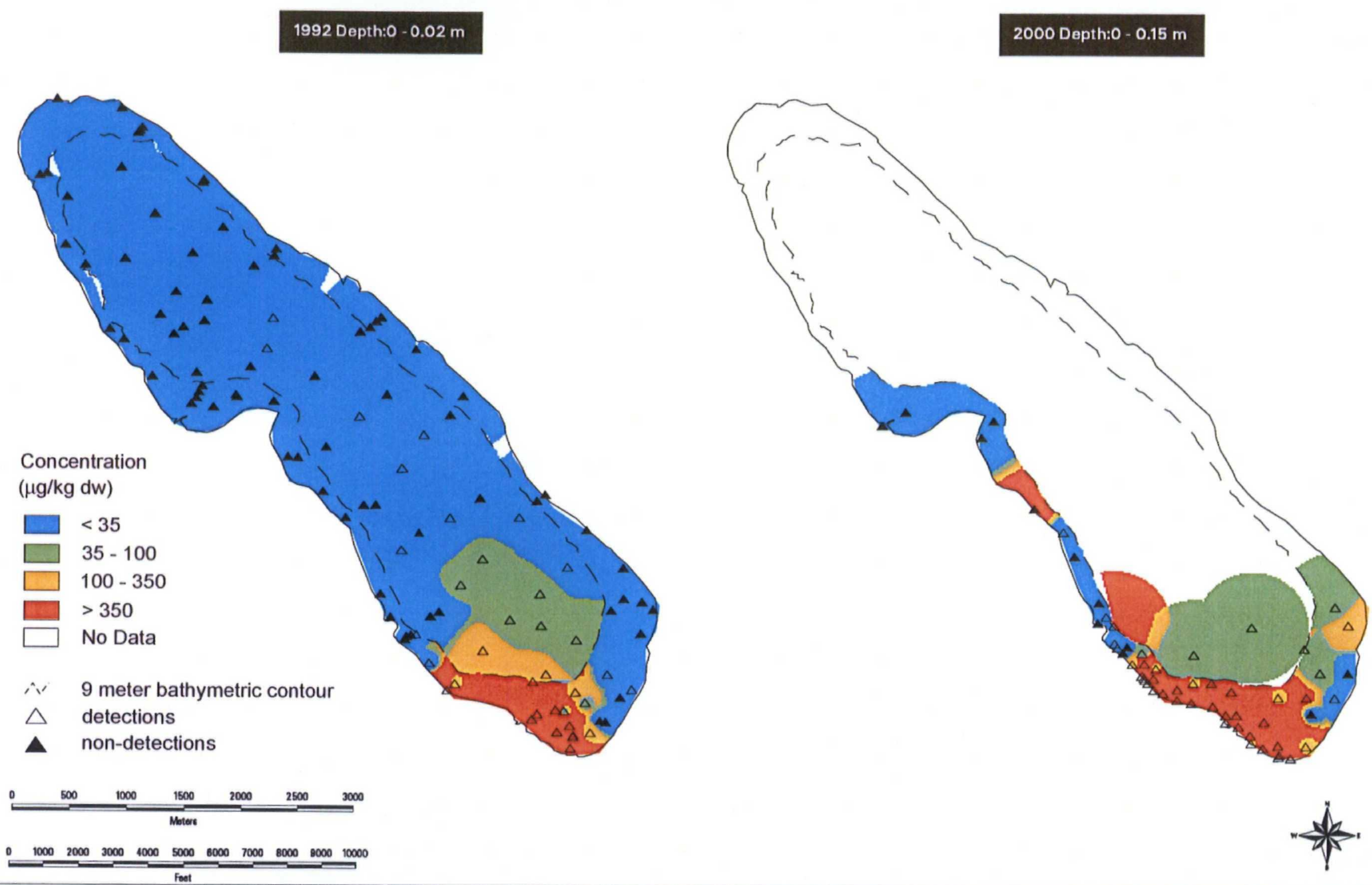
Figure 8-31. Distribution of Ethylbenzene in Surface Sediment of Onondaga Lake in 1992 and 2000

TAMS



Onondaga Lake BERA	Figure 8-32. Distribution of Toluene in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

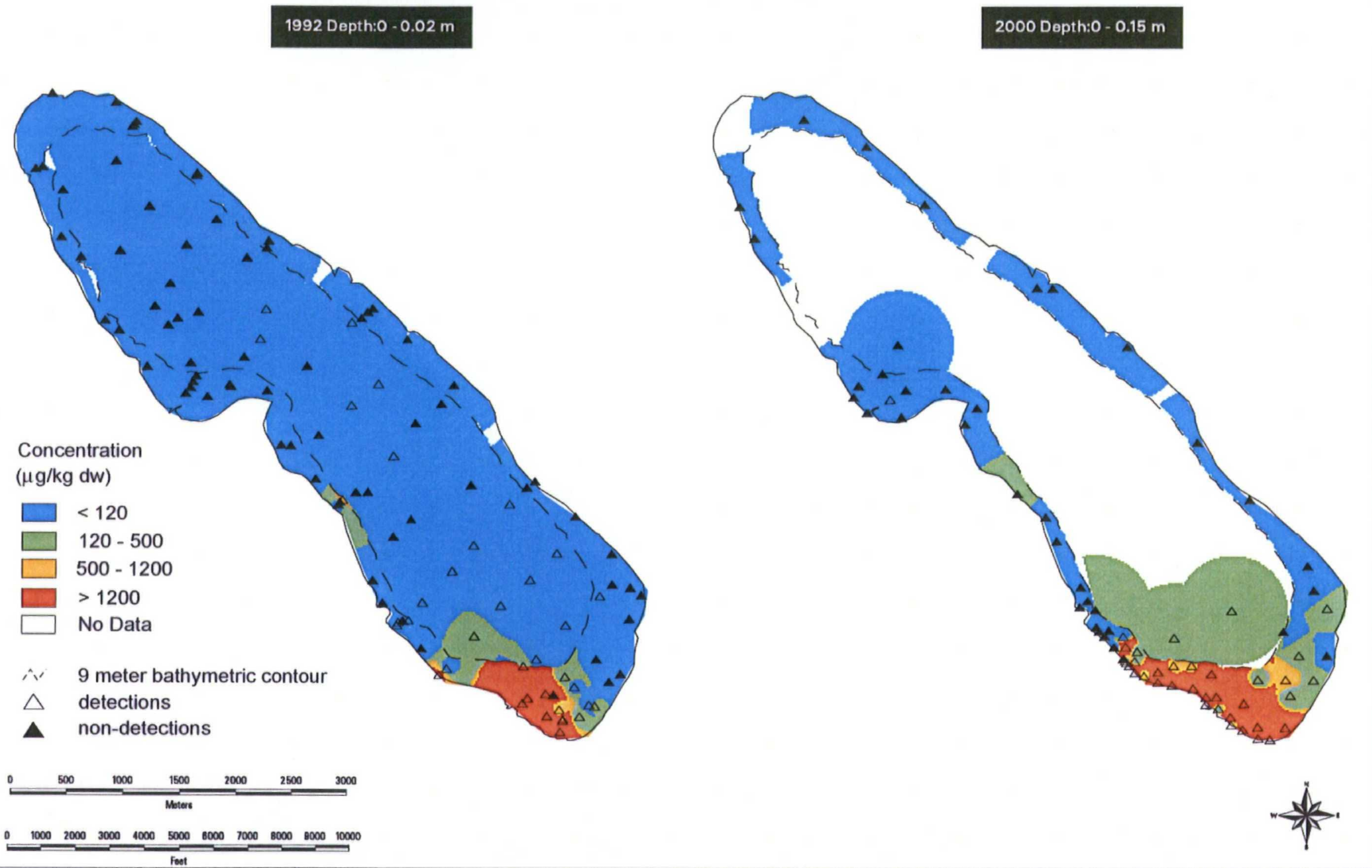
Onondaga Lake BERA	Figure 8-33. Distribution of Xylene in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



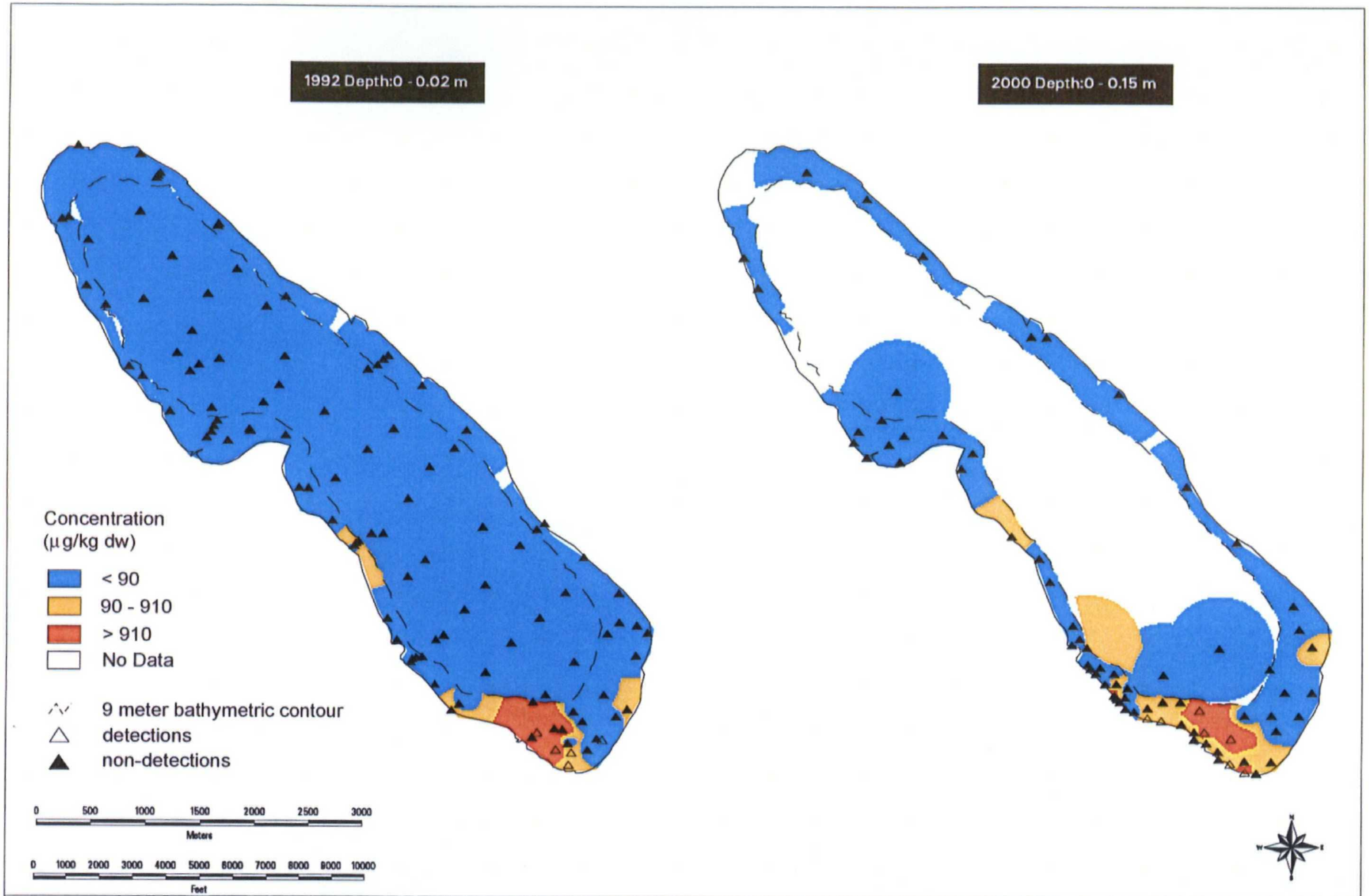
Onondaga Lake BERA

Figure 8-34. Distribution of Chlorobenzene in Surface Sediment of Onondaga Lake in 1992 and 2000

TAMS



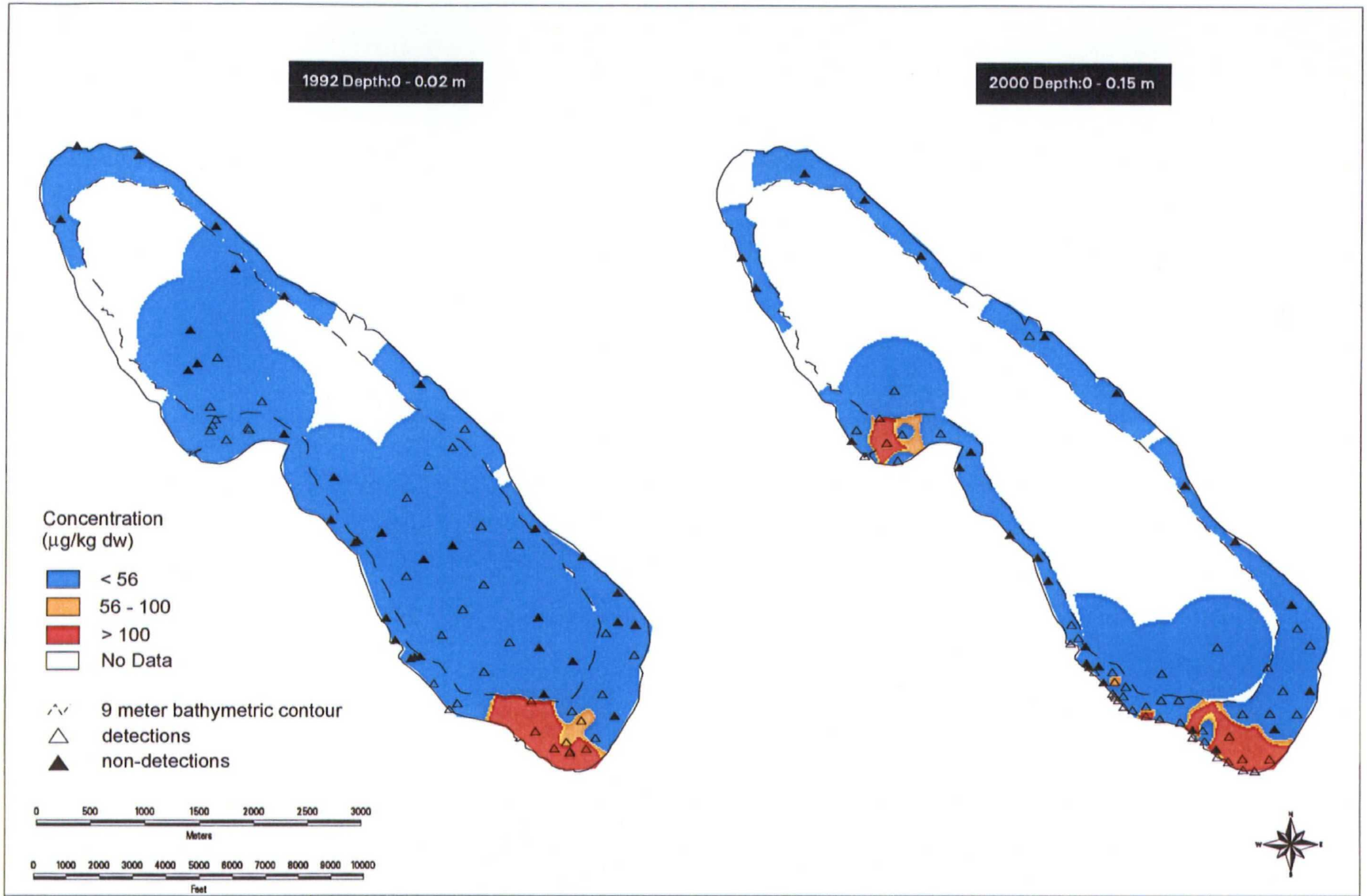
Onondaga Lake BERA	Figure 8-35. Distribution of Dichlorobenzenes in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



Onondaga Lake BERA

Figure 8-36. Distribution of Trichlorobenzenes in Surface Sediment of Onondaga Lake in 1992 and 2000

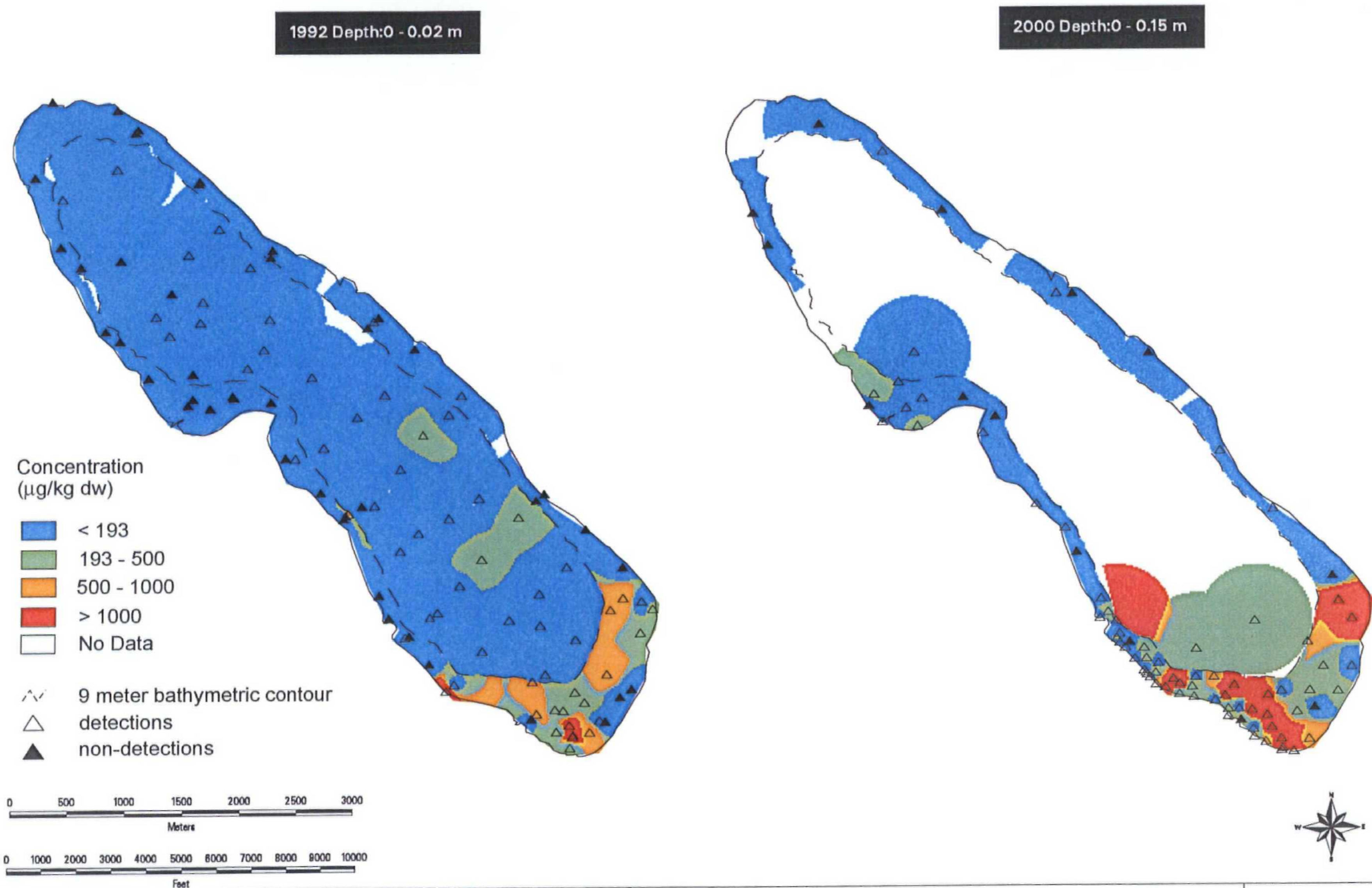
TAMS



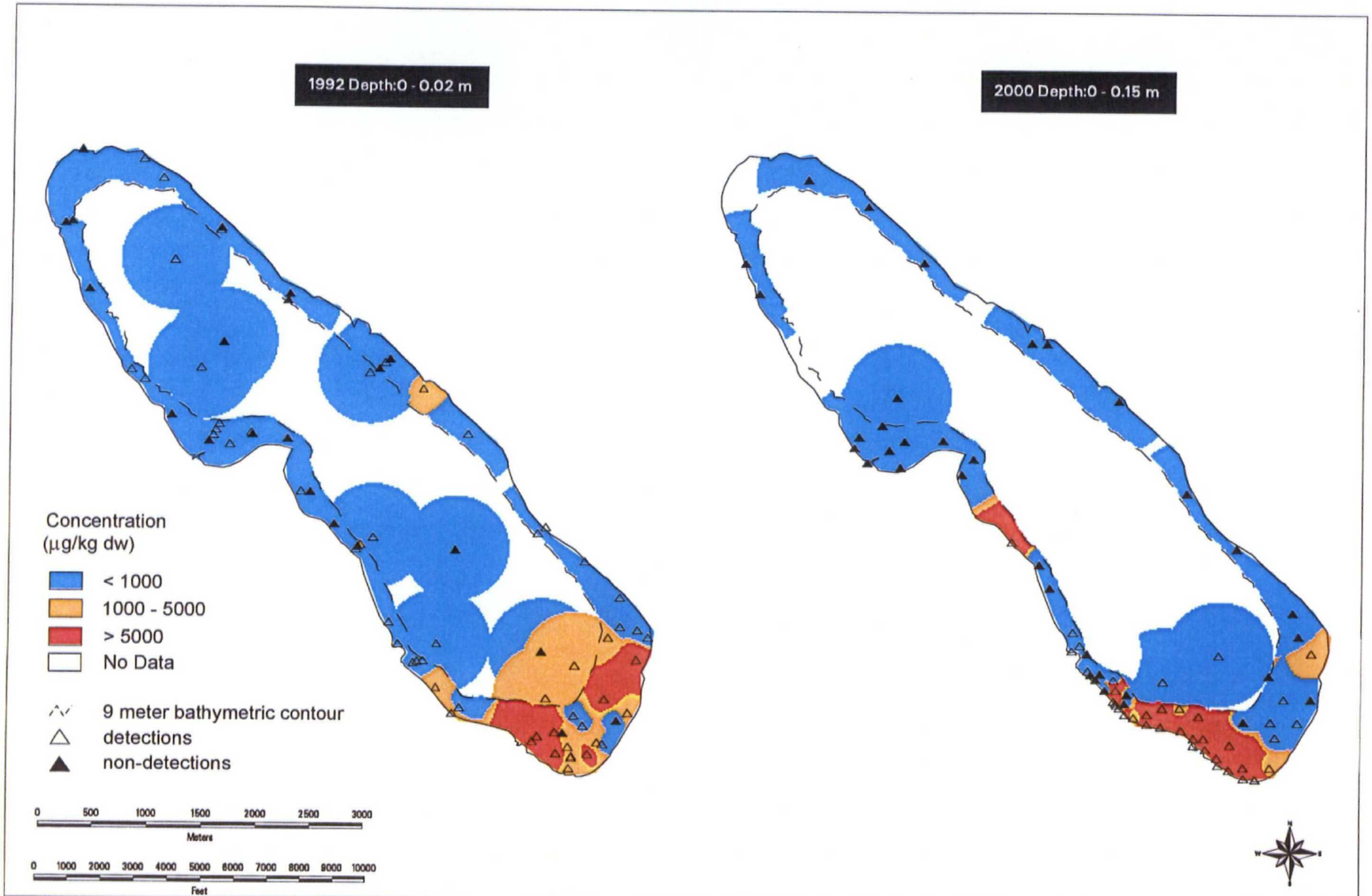
Onondaga Lake BERA

Figure 8-37. Distribution of Hexachlorobenzene in Surface Sediment of Onondaga Lake in 1992 and 2000

TAMS



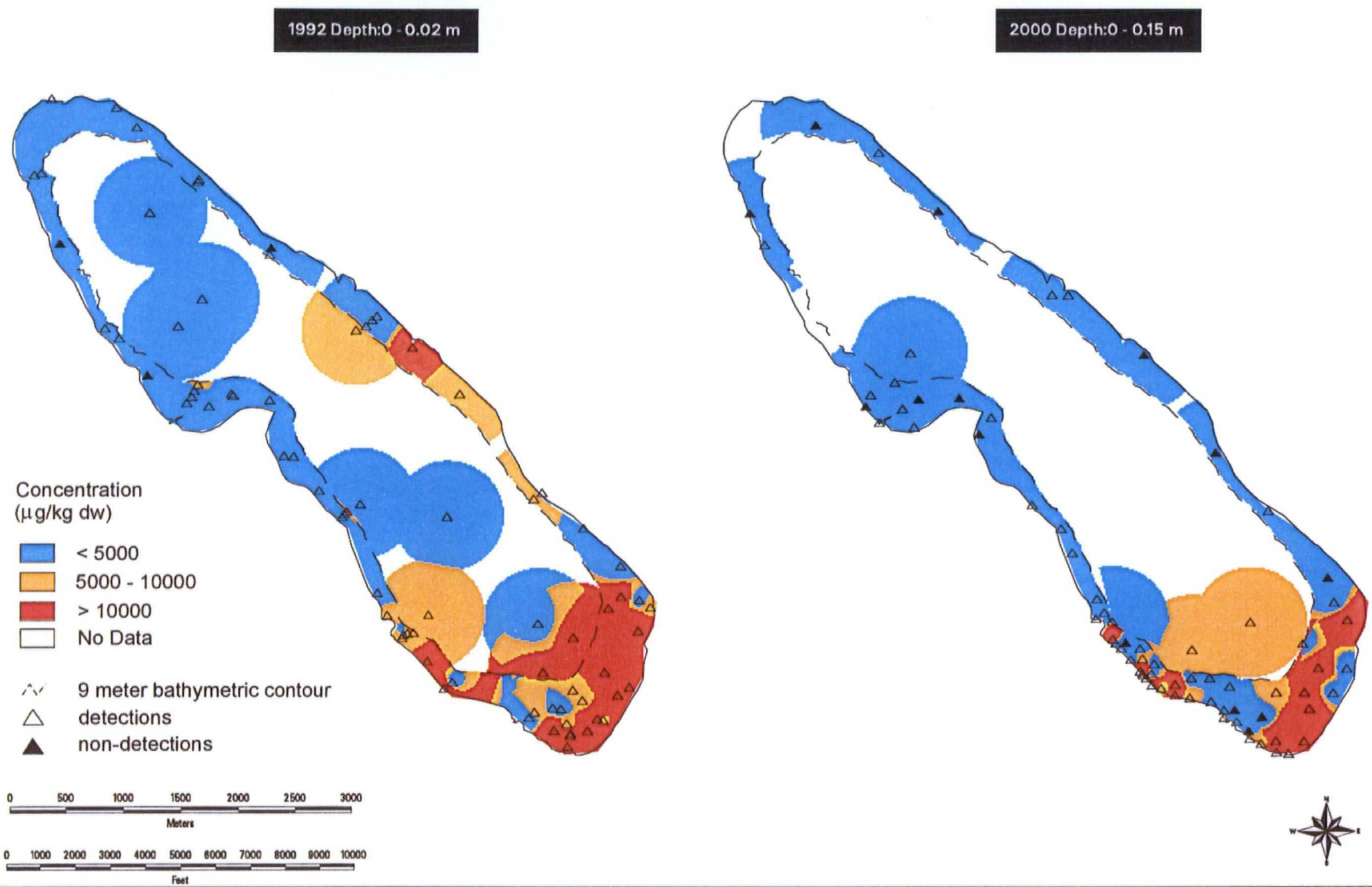
Onondaga Lake BERA	Figure 8-38. Distribution of Total PCBs in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



Onondaga Lake BERA

Figure 8-39. Distribution of LPAHs in Surface Sediment of Onondaga Lake in 1992 and 2000

TAMS

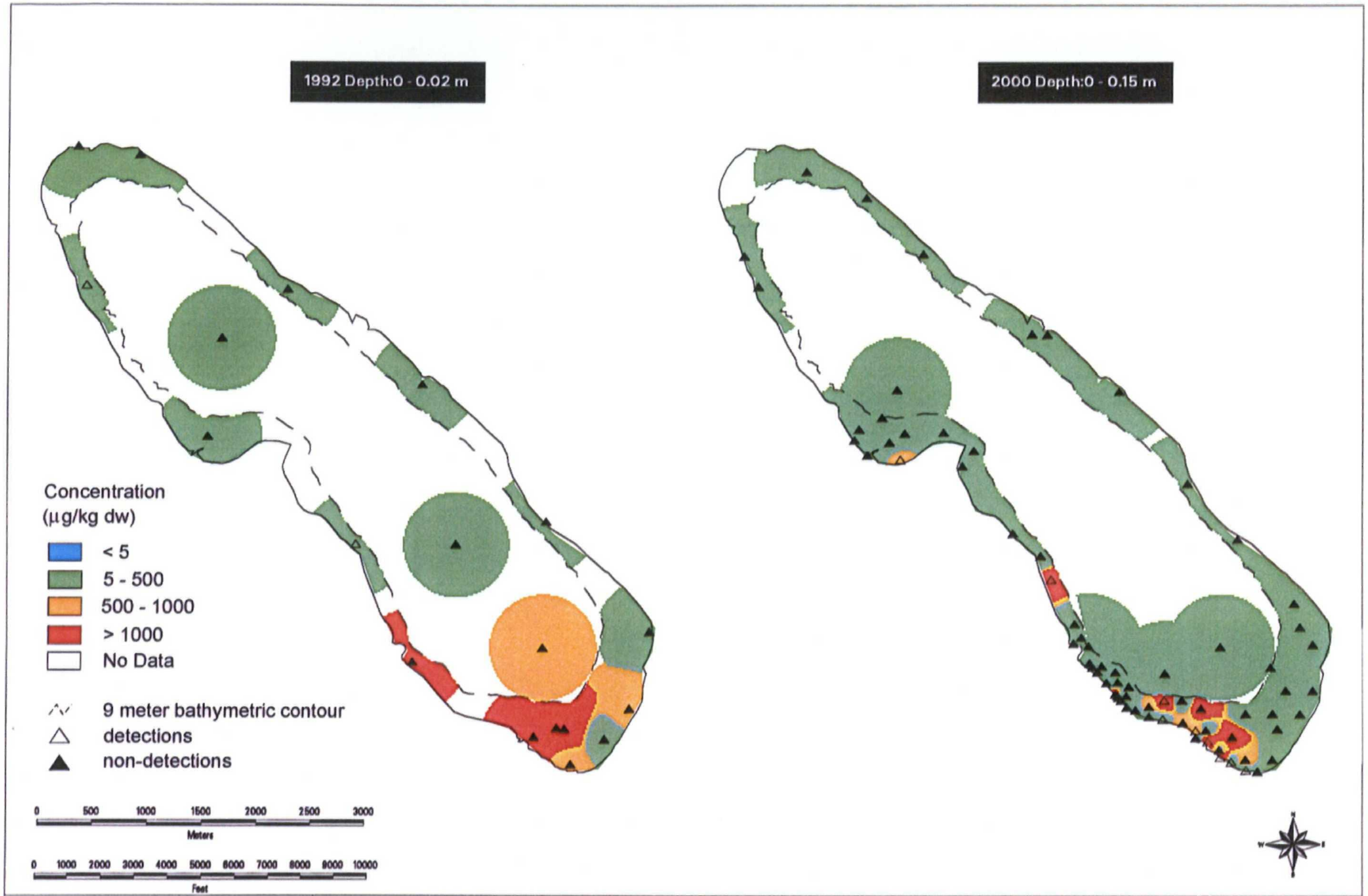


Onondaga Lake BERA	Figure 8-40. Distribution of HPAHs in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

Onondaga Lake BERA

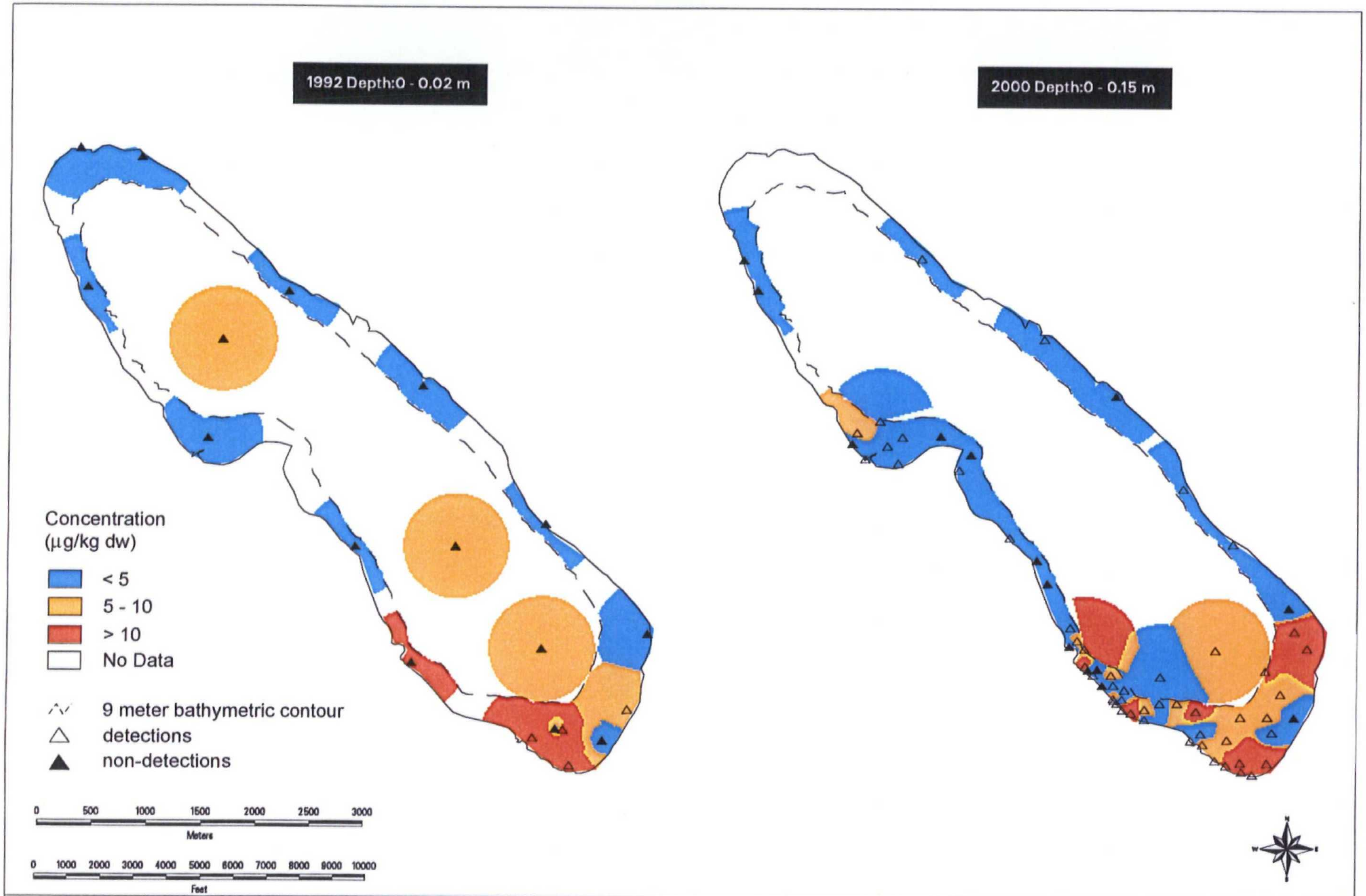
Figure 8-41. Distribution of Avian TEQ in Surface Sediment of Onondaga Lake in 2000

TAMS

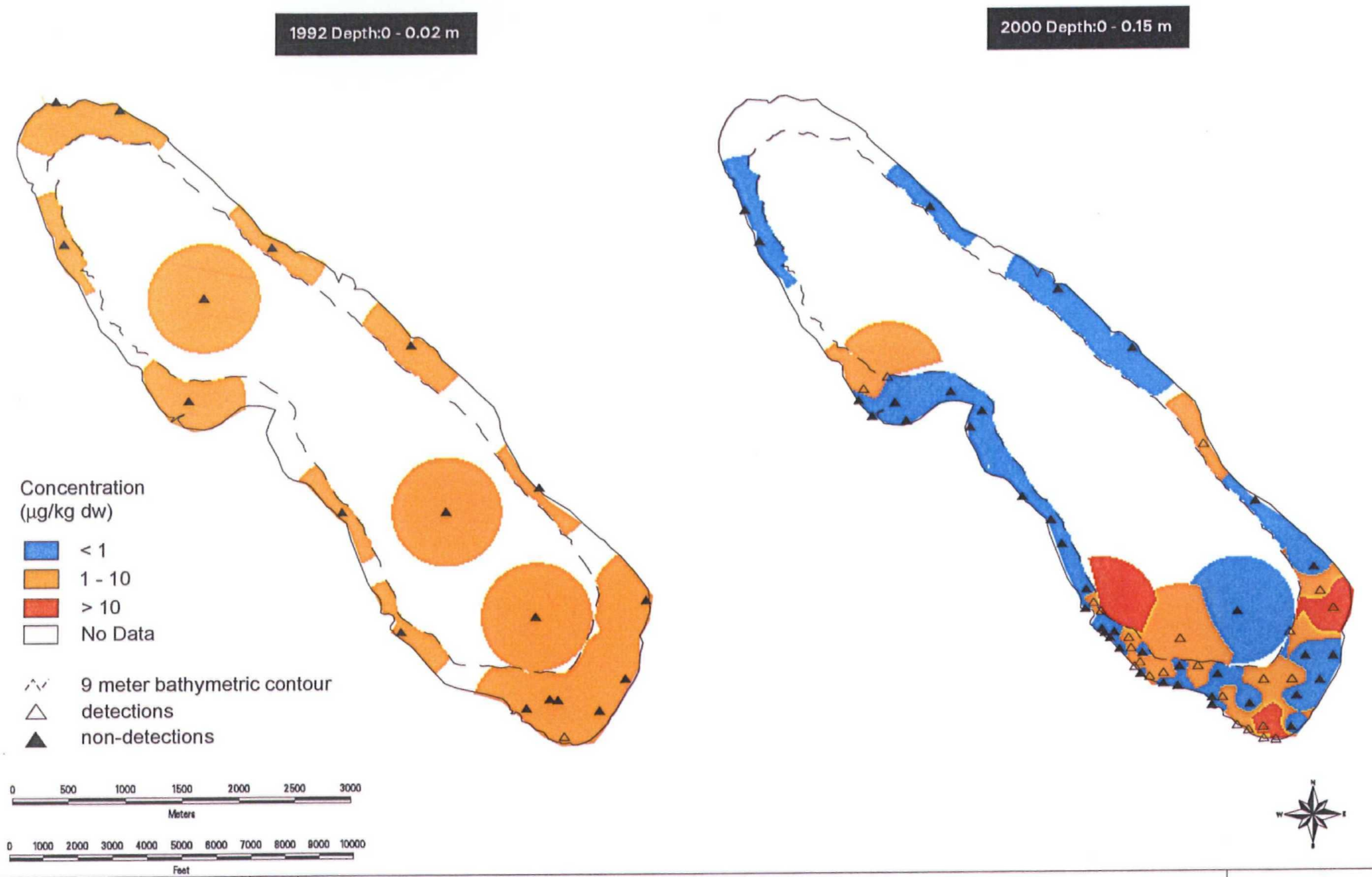


Onondaga Lake BERA	Figure 8-42. Distribution of Phenol in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

Onondaga Lake BERA	Figure 8-43. Distribution of Dibenzofuran in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

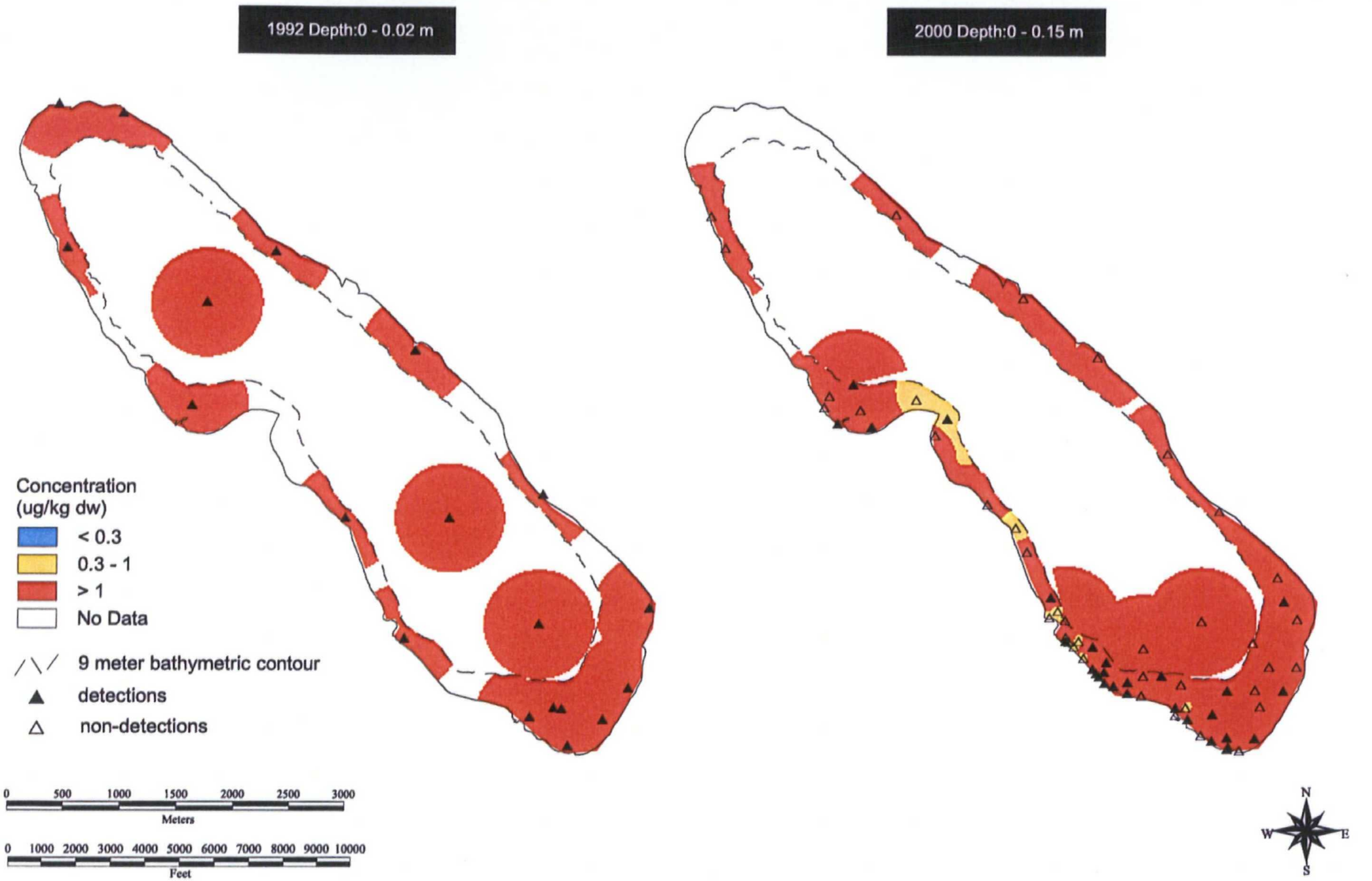


Onondaga Lake BERA	Figure 8-44. Distribution of DDT in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

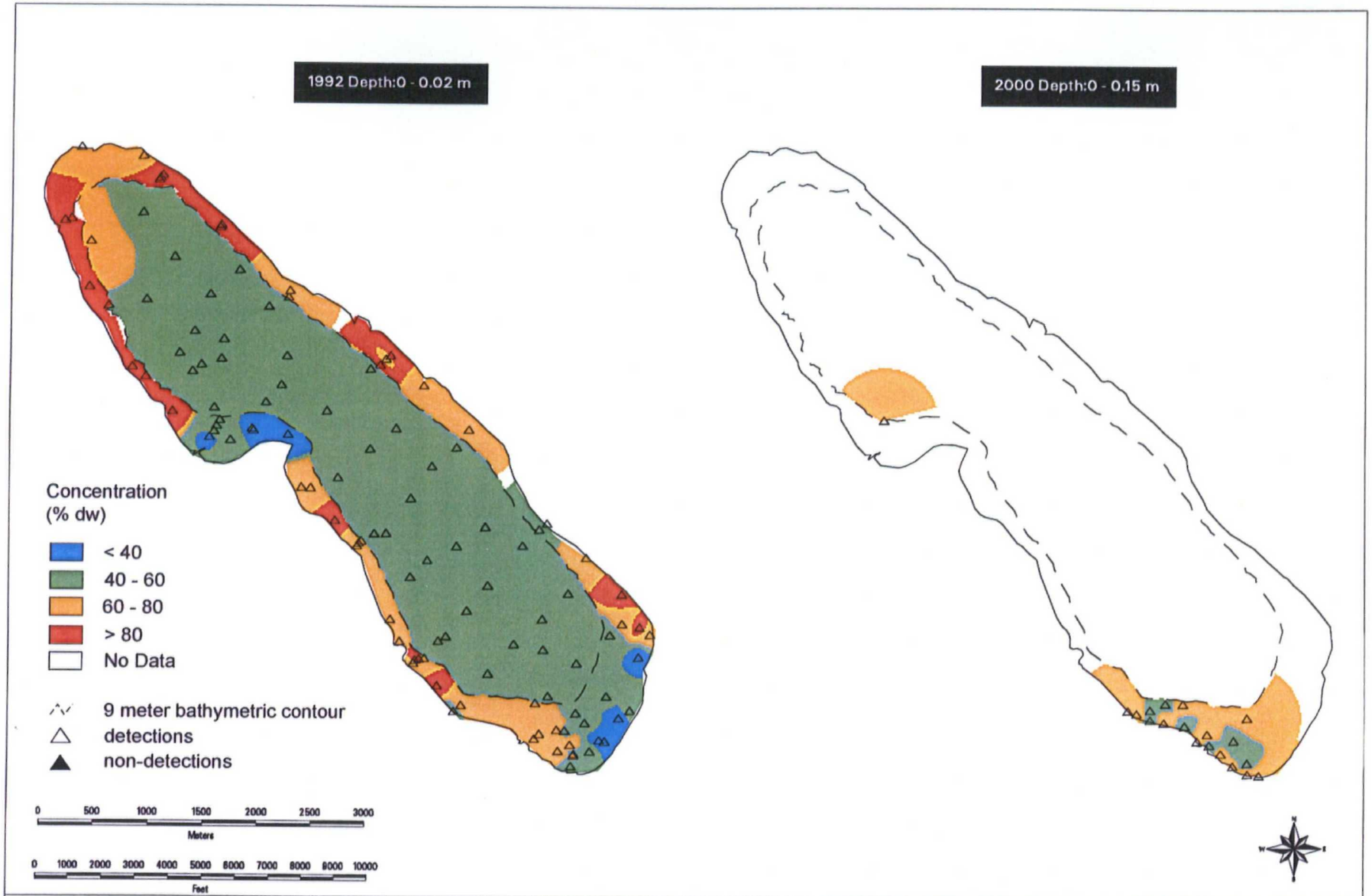


Onondaga Lake BERA	Figure 8-45. Distribution of Chlordane in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

Onondaga Lake BERA	Figure 8-46. Distribution of Dieldrin in Surface Sediments of Onondaga Lake in 2000	TAMS



Onondaga Lake BERA	Figure 8-47. Distribution of Heptachlor/Heptachlor Epoxide in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS



Onondaga Lake BERA	Figure 8-48. Distribution of Calcium Carbonate in Surface Sediment of Onondaga Lake in 1992 and 2000	TAMS

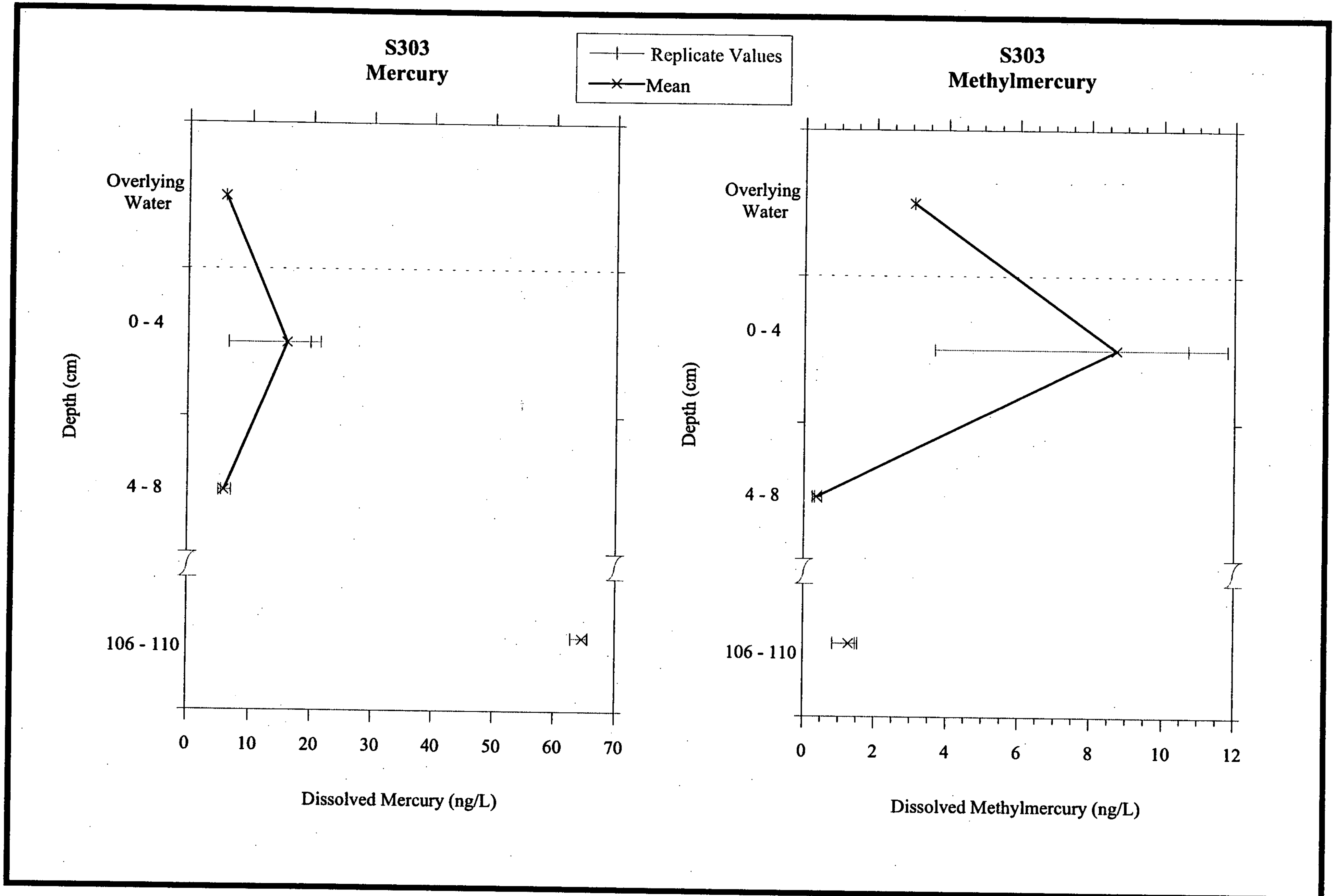


Figure 8-49
Mercury and Methylmercury in Porewater Extract at Station S303 in 2000

TAMS

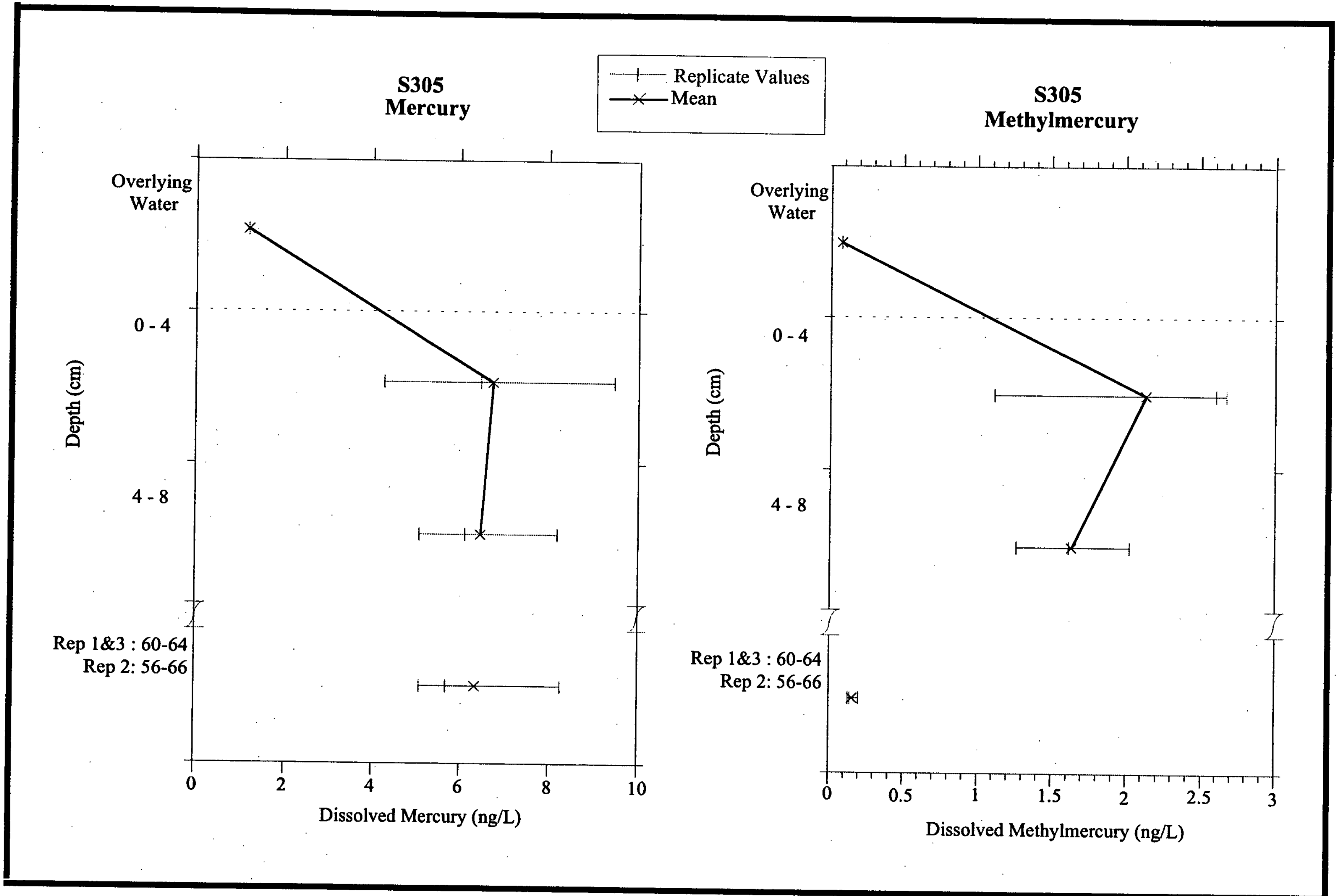


Figure 8-50
Mercury and Methylmercury in Porewater Extract at Station S305 in 2000

TAMS

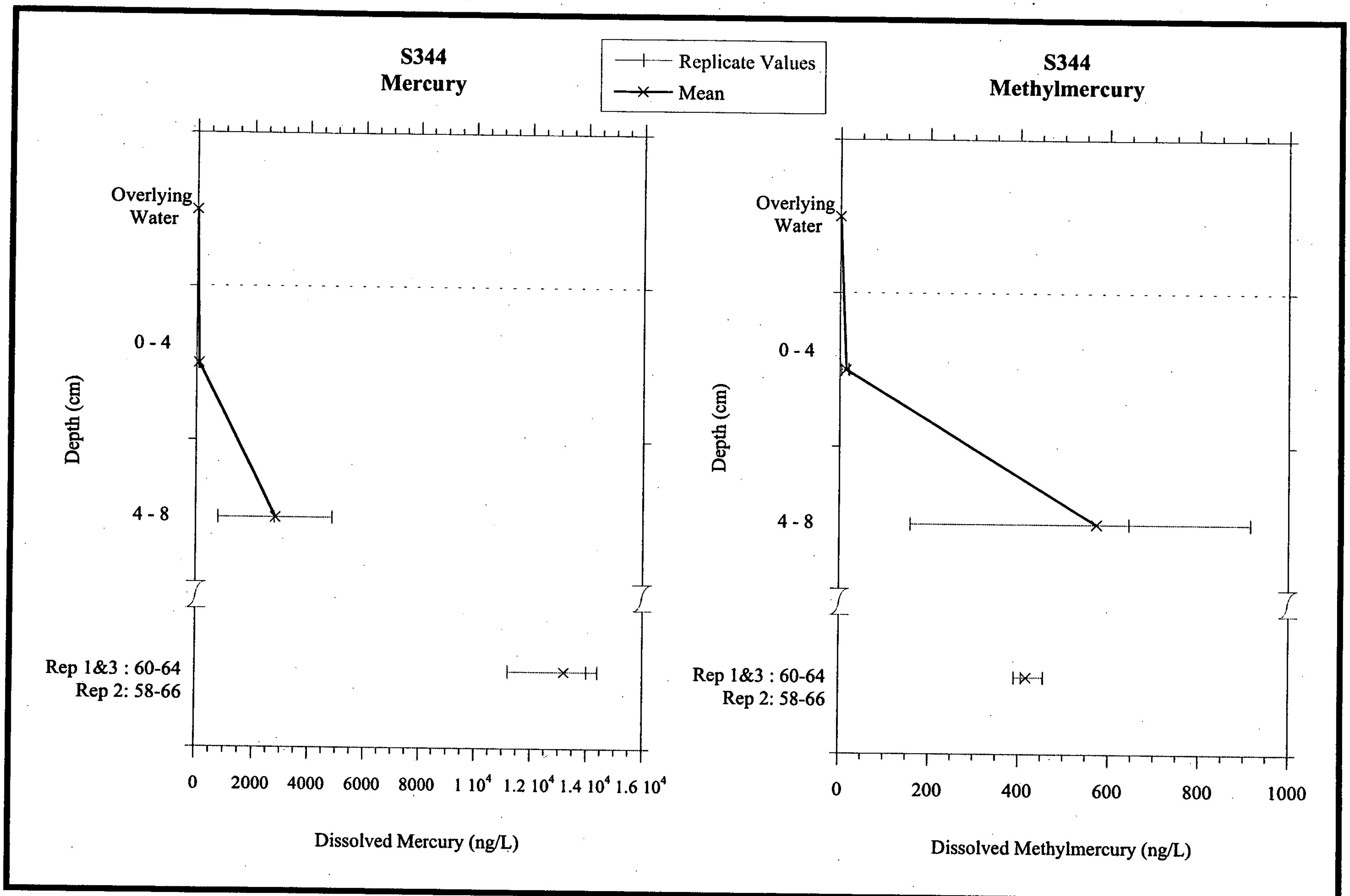


Figure 8-51
Mercury and Methylmercury in Porewater Extract at Station S344 in 2000

TAMS

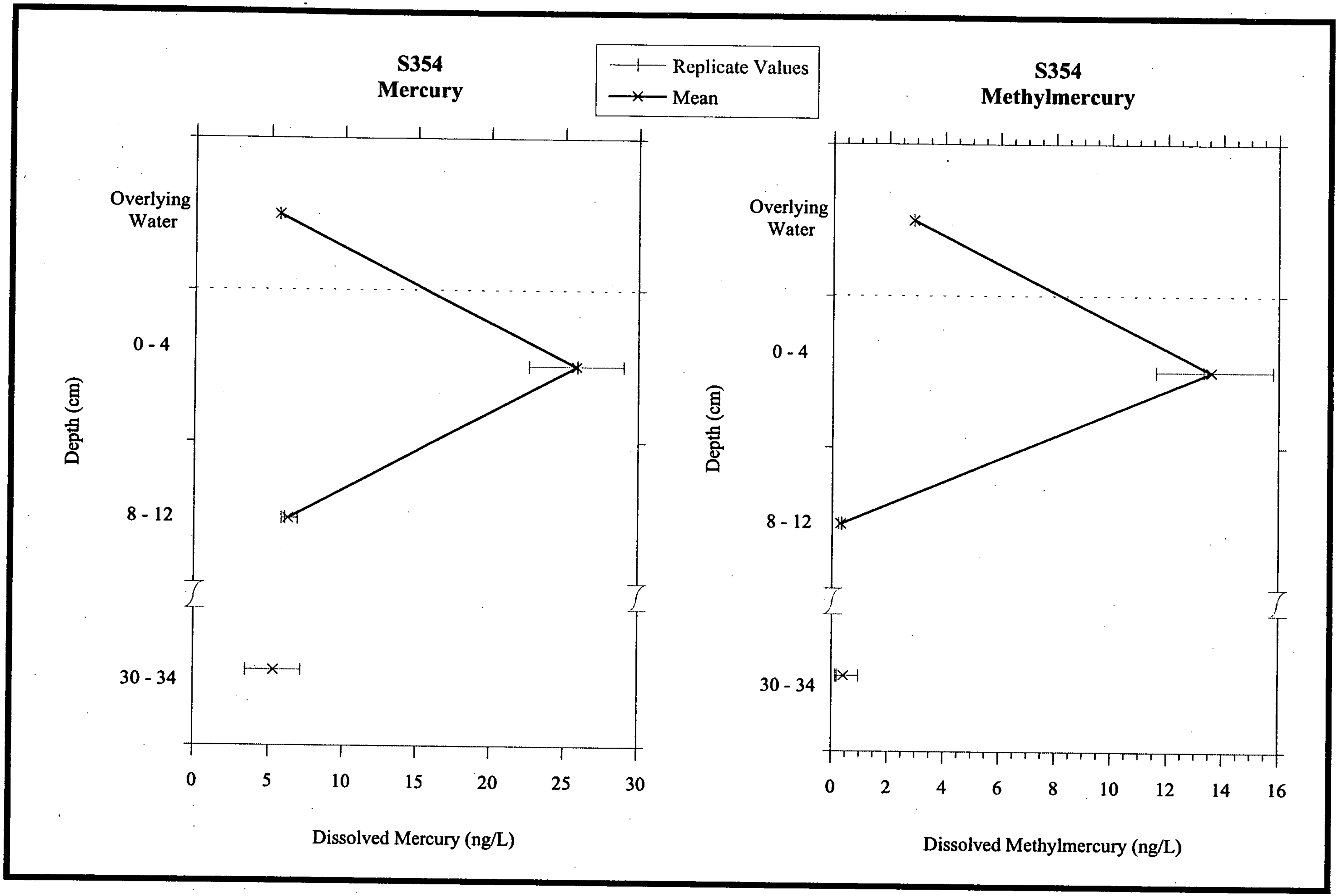


Figure 8-52
Mercury and Methylmercury in Porewater Extract at Station S354 in 2000

TAMS

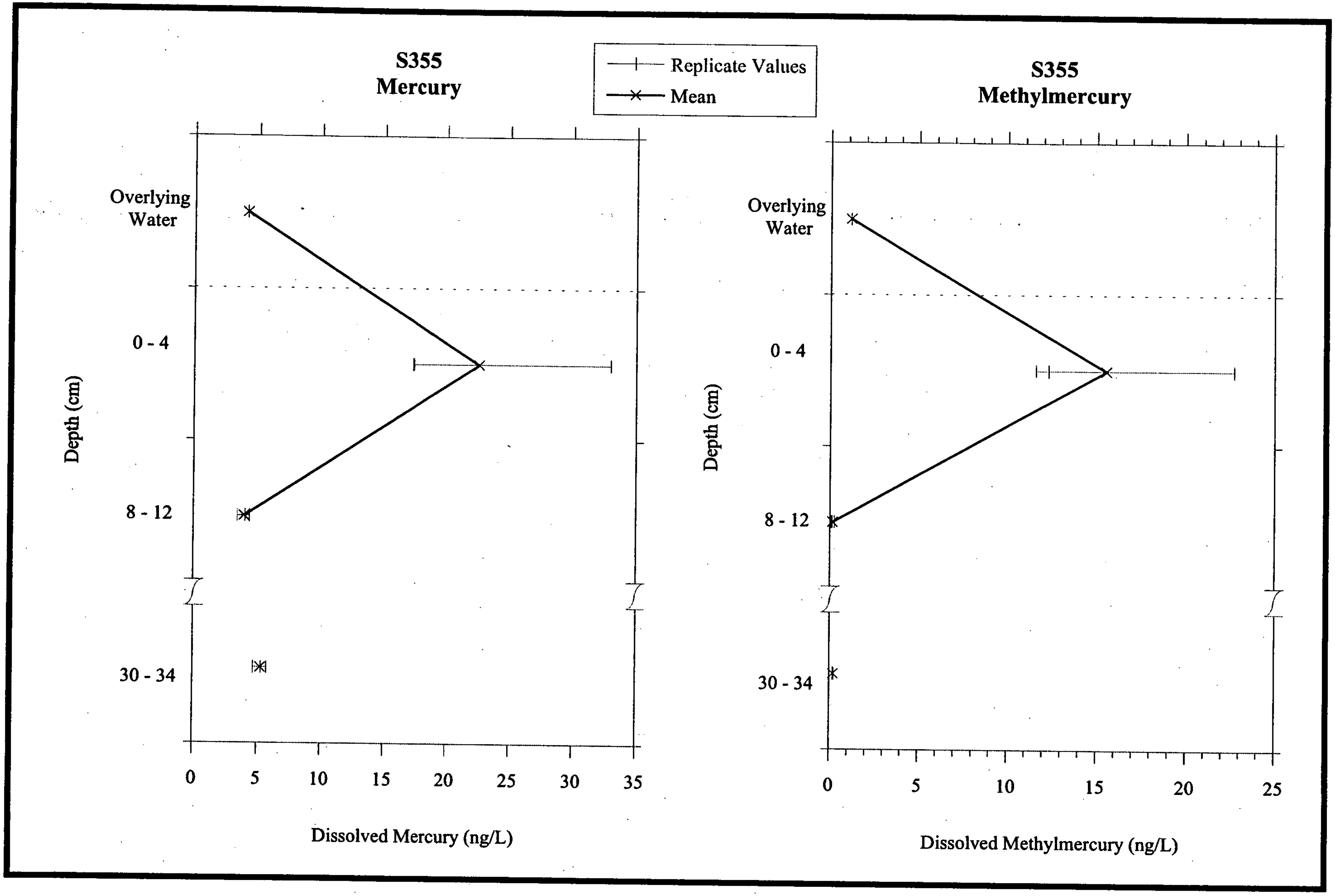


Figure 8-53
Mercury and Methylmercury in Porewater Extract at Station S355 in 2000

TAMS

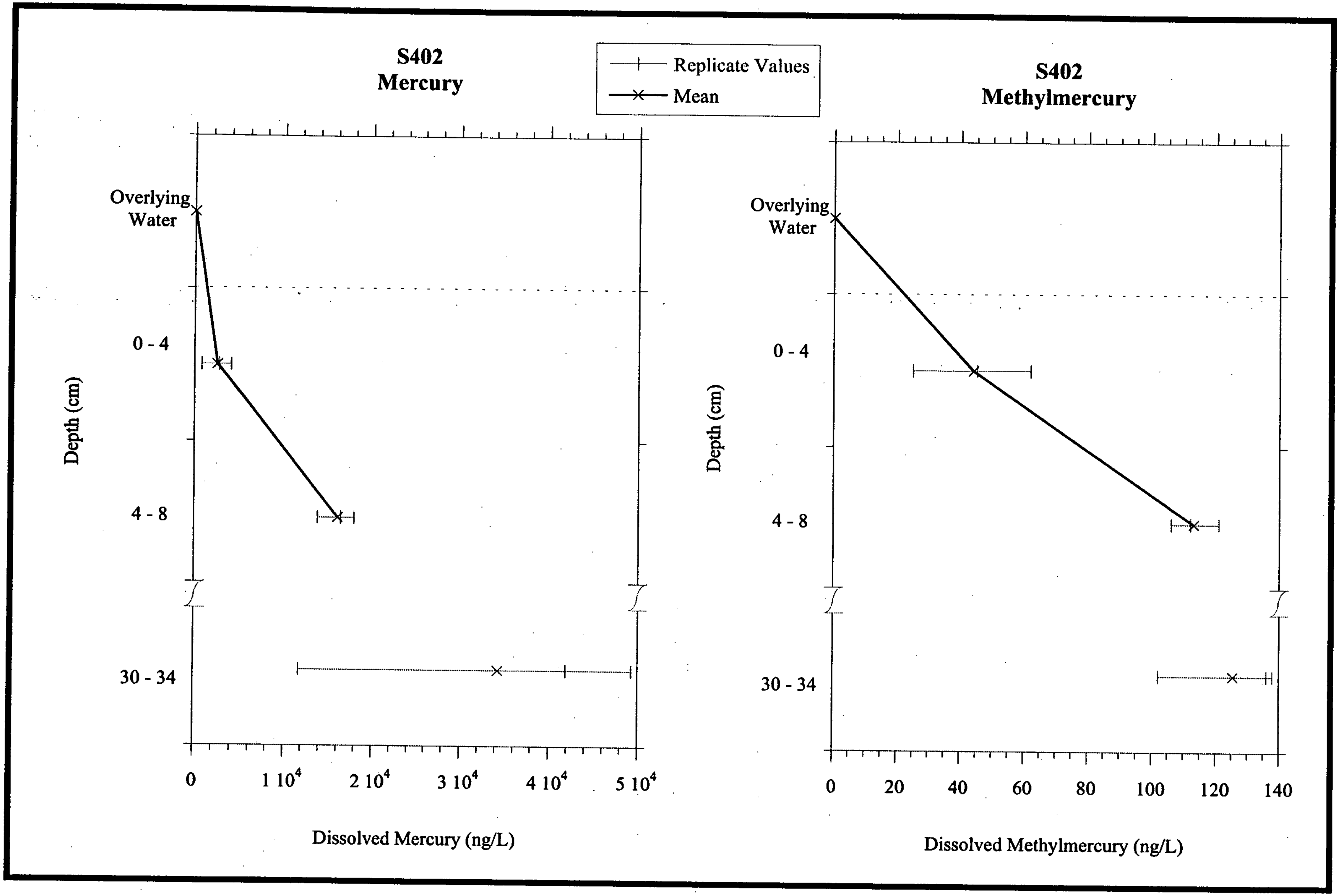


Figure 8-54
Mercury and Methylmercury in Porewater Extract at Station S402 in 2000

TAMS

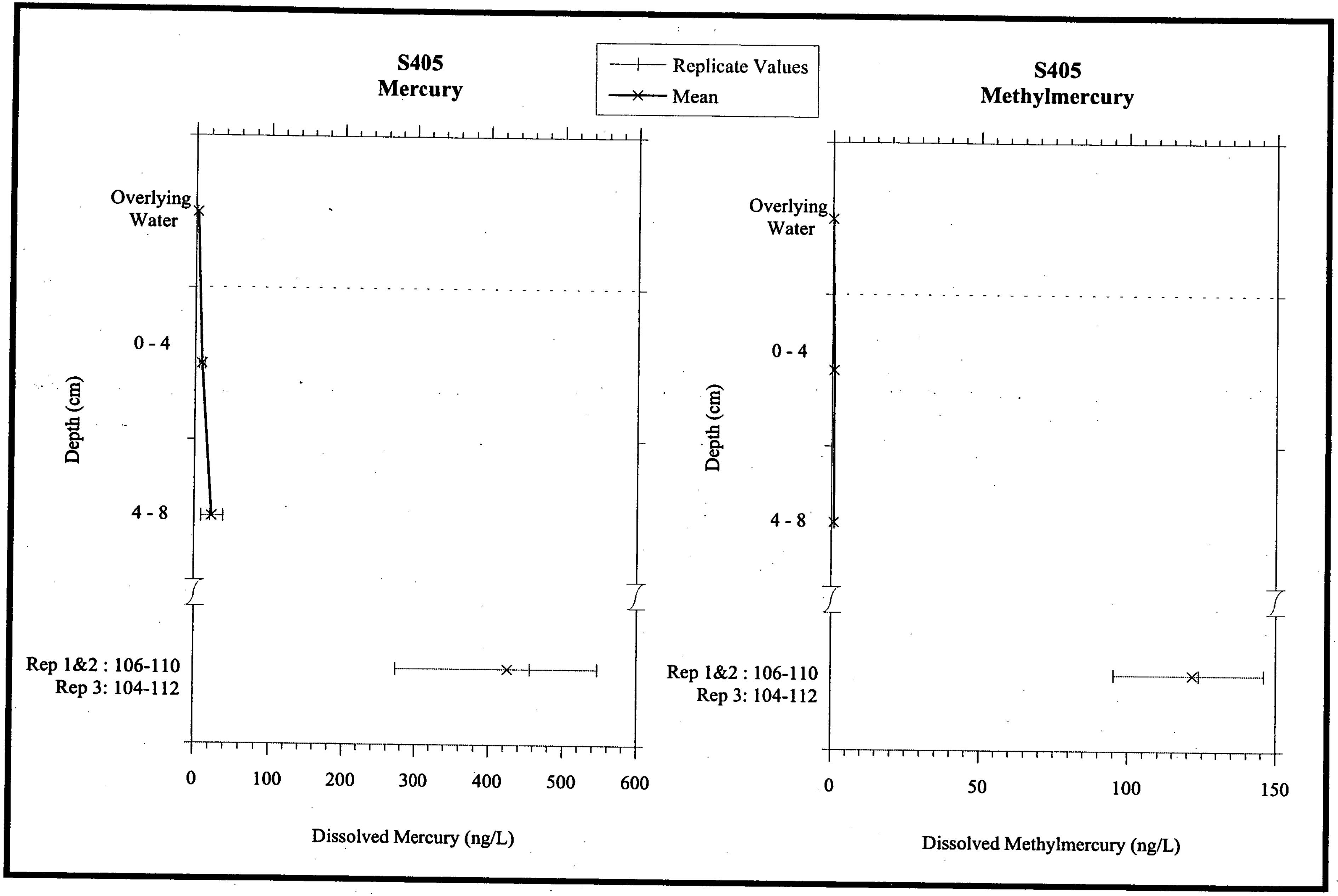


Figure 8-55
Mercury and Methylmercury in Porewater Extract at Station S405 in 2000

TAMS

Table 8-1. Summary of Concentrations of Metals Other than Mercury in Tributary and Onondaga Lake Water and Metro Discharge in 1992 and 1999

			Metals (µg/L)													
		No. of	Barium			Cadmium			Chromium			Copper				
Station	Location	Samples	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.		
Tributaries/Metro Outfall–Base Flow, 1992																
W3	Metro Outfall	4	0	--	--	0	--	--	1	2.5	2.5	4	8.0	12		
W4	Onondaga Creek	5	0	--	--	0	--	--	0	--	--	3	4.2	6.4		
W5	Harbor Brook	7	0	--	--	0	--	--	1	4.5	4.5	1	1.6	1.6		
W6	Ley Creek	4	0	--	--	0	--	--	3	3.2	4.3	3	3.4	4.1		
W7	East Flume	10	0	--	--	0	--	--	3	6.2	11	9	7.5	15		
W8	Tributary 5A	7	0	--	--	0	--	--	7	13	28	6	7.7	10		
W9	Bloody Brook	1	0	--	--	0	--	--	0	--	--	1	1.9	1.9		
W10	Ninemile Creek	3	0	--	--	0	--	--	0	--	--	0	--	--		
W11	Sawmill Creek	1	0	--	--	0	--	--	0	--	--	0	--	--		
W12	Lake Outlet	3	0	--	--	0	--	--	1	18	18	0	--	--		
Tributaries/Metro Outfall–Intermediate Flow, 1992																
W3	Metro Outfall	1	0	--	--	0	--	--	0	--	--	1	4.2	4.2		
W4	Onondaga Creek	9	0	--	--	0	--	--	1	2.2	2.2	6	6.2	11		
W5	Harbor Brook	3	0	--	--	0	--	--	2	5.4	5.7	1	6.7	6.7		
W6	Ley Creek	7	0	--	--	0	--	--	1	3.9	3.9	5	7.4	13		
W7	East Flume	5	0	--	--	0	--	--	3	5.0	8.3	5	9.0	18		
W8	Tributary 5A	3	0	--	--	1	2.4	2.4	3	69	119	3	20	30		
W10	Ninemile Creek	9	0	--	--	0	--	--	2	9.9	12	6	6.4	16		
W12	Lake Outlet	10	0	--	--	0	--	--	0	--	--	0	--	--		
Tributaries/Metro Outfall–High Flow, 1992																
W3	Metro Outfall	7	0	--	--	0	--	--	3	3.4	4.3	7	8.7	13		
W4	Onondaga Creek	5	0	--	--	0	--	--	0	--	--	3	4.8	9.5		
W5	Harbor Brook	8	0	--	--	0	--	--	4	7.1	9.1	8	22	48		
W6	Ley Creek	5	0	--	--	0	--	--	3	12	19	5	23	58		
W7	East Flume	4	0	--	--	0	--	--	0	--	--	4	4.5	7.5		
W8	Tributary 5A	9	0	--	--	1	3.2	3.2	9	200	560	9	49	125		
W9	Bloody Brook	1	0	--	--	1	17	17	1	12	12	1	42	42		
W10	Ninemile Creek	5	0	--	--	1	2.1	2.1	2	4.7	6.4	5	6.5	16		
W11	Sawmill Creek	1	0	--	--	0	--	--	0	--	--	1	4.7	4.7		
W12	Lake Outlet	17	0	--	--	0	--	--	0	--	--	7	2.7	4.9		

Table 8-1. (cont.)

			Metals (µg/L)											
		No. of	Barium			Cadmium			Chromium			Copper		
Station	Location	Samples	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.
Onondaga Lake, 1992														
W1	South Basin	66	2	73	77	1	2.9	2.9	11	2.8	5.3	18	6.6	51
W2	North Basin	44	2	71	76	2	2.7	2.7	5	2.7	4.2	12	1.9	3
Onondaga Lake, 1999														
W1	South Basin	1	0	--	--	0	--	--	1	3.5	3.5	0	--	--
W2	North Basin	1	0	--	--	0	--	--	1	3.7	3.7	0	--	--
W50	Willis Lakeshore Exposure Area	1	0	--	--	0	--	--	1	3.9	3.9	0	--	--
W51	Observed Fish Area	1	0	--	--	0	--	--	1	3.7	3.7	0	--	--
W52	Access from Fairgrounds	1	0	--	--	0	--	--	1	3.5	3.5	0	--	--
W53	Beach Access	2	0	--	--	0	--	--	2	3.1	3.5	0	--	--
W54	Lake Park Lakeland	1	0	--	--	0	--	--	1	3.1	3.1	0	--	--
W55	Harbor Brook	1	0	--	--	0	--	--	1	3.8	3.8	0	--	--
W56	Park/Picnic Area/Playground	1	0	--	--	0	--	--	1	3.6	3.6	0	--	--
W57	Boat Ramp (Liverpool)	1	0	--	--	0	--	--	1	3.2	3.2	0	--	--
W58	Lake Park Galeville	1	0	--	--	0	--	--	1	3.2	3.2	0	--	--

Table 8-1. (cont.)

			Metals (µg/L)													
		No. of	Lead			Manganese			Nickel			Zinc				
Station	Location	Samples	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.		
Tributaries/Metro Outfall–Base Flow, 1992																
W3	Metro Outfall	4	1	1.1	1.1	0	--	--	1	9.3	9.3	3	33	42		
W4	Onondaga Creek	5	2	2.4	3.5	0	--	--	0	--	--	4	26	51		
W5	Harbor Brook	7	1	1.7	1.7	0	--	--	1	9.1	9.1	4	5.0	6.4		
W6	Ley Creek	4	3	4.6	7.4	0	--	--	3	9.0	9.5	3	17	22		
W7	East Flume	10	5	5.7	11	0	--	--	3	9.0	9.5	3	17	22		
W8	Tributary 5A	7	5	1.6	2.1	0	--	--	0	--	--	9	89	196		
W9	Bloody Brook	1	0	--	--	0	--	--	0	--	--	0	--	--		
W10	Ninemile Creek	3	0	--	--	0	--	--	3	7.0	7.7	2	4.5	5.6		
W11	Sawmill Creek	1	0	--	--	0	--	--	1	17	17	0	--	--		
W12	Lake Outlet	3	0	--	--	0	--	--	1	10	10	1	14	14		
Tributaries/Metro Outfall–Intermediate Flow, 1992																
W3	Metro Outfall	1	1	4.8	4.8	0	--	--	0	--	--	1	40	40		
W4	Onondaga Creek	9	6	7.3	17	0	--	--	0	--	--	8	27	85		
W5	Harbor Brook	3	2	4.3	6.7	0	--	--	2	6.3	6.5	3	15	29		
W6	Ley Creek	7	7	4.7	8.6	0	--	--	2	7.9	8.7	7	30	43		
W7	East Flume	5	5	6.9	28	0	--	--	0	--	--	5	122	179		
W8	Tributary 5A	3	3	9.2	14	0	--	--	9	140	327	3	76	117		
W10	Ninemile Creek	9	6	7.2	22	0	--	--	3	61	93	6	33	88		
W12	Lake Outlet	10	1	1.1	1.1	0	--	--	7	66	115	8	7.0	13		
Tributaries/Metro Outfall–High Flow, 1992																
W3	Metro Outfall	7	5	1.7	2.2	0	--	--	7	9.3	15	7	43	71		
W4	Onondaga Creek	5	3	6.8	16	0	--	--	0	--	--	5	13	38		
W5	Harbor Brook	8	6	35	63	0	--	--	3	16	19	8	97	188		
W6	Ley Creek	5	5	31	95	0	--	--	0	--	--	5	84	182		
W7	East Flume	4	3	1.6	2.0	0	--	--	0	--	--	4	109	130		
W8	Tributary 5A	9	7	33	55	0	--	--	0	--	--	8	119	259		
W9	Bloody Brook	1	1	44	44	0	--	--	4	10	17	1	201	201		
W10	Ninemile Creek	5	3	9.3	20	0	--	--	0	--	--	3	43	72		
W11	Sawmill Creek	1	1	3.5	3.5	0	--	--	1	9.1	9.1	1	27	27		
W12	Lake Outlet	17	2	4.2	5.8	0	--	--	1	11	11	9	18	47		

Table 8-1. (cont.)

			Metals (µg/L)											
		No. of	Lead			Manganese			Nickel			Zinc		
Station	Location	Samples	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.	No. Det.	Mean Conc.	Max. Conc.
Onondaga Lake, 1992														
W1	South Basin	66	14	1.7	2.8	66	190	868	10	9.7	15	46	17	143
W2	North Basin	44	10	2.4	7.7	42	173	880	2	5.3	5.3	31	12	42
Onondaga Lake, 1999														
W1	South Basin	1	0	--	--	34	143.1	601	1	3.6	3.6	0	--	--
W2	North Basin	1	0	--	--	21	186.2	577	1	3.7	3.7	0	--	--
W50	Willis Lakeshore Exposure Area	1	1	3.1	3.1	1	21	21	1	4.6	4.6	0	--	--
W51	Observed Fish Area	1	0	--	--	1	27	27	1	4.7	4.7	0	--	--
W52	Access from Fairgrounds	1	0	--	--	2	51	87	1	4.2	4.2	0	--	--
W53	Beach Access	2	0	--	--	1	15	15	2	4.0	4.0	0	--	--
W54	Lake Park Lakeland	1	0	--	--	2	58	102	1	3.7	3.7	0	--	--
W55	Harbor Brook	1	0	--	--	1	31	31	1	4.2	4.2	0	--	--
W56	Park/Picnic Area/Playground	1	0	--	--	1	19	19	1	3.5	3.5	0	--	--
W57	Boat Ramp (Liverpool)	1	0	--	--	1	16	16	1	3.9	3.9	0	--	--
W58	Lake Park Galeville	1	0	--	--	2	63	110	1	4.0	4.0	0	--	--

Source: Onondaga Lake Database

Notes: -- not detected

1. Detailed information for each water sample is presented in Appendix B.
2. Mean concentration was calculated using only detected values.
3. Barium was only measured in two samples in 1992.
4. All manganese detections were in 1999.

Table 8-2. Summary of Concentrations of Benzene, Toluene, Ethylbenzene and Xylenes (BTEX) in Tributary and Onondaga Lake Water and Metro Discharge in 1992 and 1999

Station	Location	No. of Samples	Benzene No. of Detects	Benzene Max. Conc. (μg/L)	Toluene No. of Detects	Toluene Max. Conc. (μg/L)	Xylenes No. of Detects	Xylenes Max. Conc. (μg/L)	Ethylbenzene No. of Detects	Ethylbenzene Max. Conc. (μg/L)
Tributaries/Metro Outfall–Base Flow, 1992										
W3	Metro Outfall	3	0	--	1	3.1	0	--	0	--
W4	Onondaga Creek	4	0	--	0	--	0	--	0	--
W5	Harbor Brook	7	4	1.7	5	2.6	4	3.6	0	--
W6	Ley Creek	3	0	--	0	--	0	--	0	--
W7	East Flume	10	1	15	1	2.5	1	1.4	0	--
W8	Tributary 5A	7	3	34	3	4.2	4	2.2	0	--
W9	Bloody Brook	1	0	--	0	--	0	--	0	--
W10	Ninemile Creek	3	0	--	0	--	0	--	0	--
W11	Sawmill Creek	1	0	--	0	--	0	--	0	--
W12	Lake Outlet	2	0	--	0	--	0	--	0	--
Tributaries/Metro Outfall–Intermediate Flow, 1992										
W3	Metro Outfall									
W4	Onondaga Creek	7	0	--	0	--	0	--	0	--
W5	Harbor Brook	3	0	--	0	--	0	--	0	--
W6	Ley Creek	5	0	--	0	--	0	--	1	1.7
W7	East Flume	5	0	--	0	--	0	--	0	--
W8	Tributary 5A	3	0	--	0	--	0	--	0	--
W10	Ninemile Creek	6	0	--	0	--	0	--	0	--
W12	Lake Outlet	7	0	--	0	--	0	--	0	--
Tributaries/Metro Outfall–High Flow, 1992										
W3	Metro Outfall	8	0	--	0	--	0	--	0	--
W4	Onondaga Creek	5	0	--	0	--	0	--	0	--
W5	Harbor Brook	9	0	--	1	2.1	1	1.7	0	--
W6	Ley Creek	4	0	--	0	--	0	--	0	--
W7	East Flume	4	0	--	0	--	3	1.8	0	--
W8	Tributary 5A	9	1	60	1	5.1	2	2.4	0	--
W9	Bloody Brook	1	0	--	0	--	0	--	0	--
W10	Ninemile Creek	5	0	--	0	--	0	--	0	--
W11	Sawmill Creek	1	0	--	0	--	0	--	0	--
W12	Lake Outlet	16	0	--	0	--	0	--	0	--
Onondaga Lake, 1992										
W1	South Basin	60	0	--	0	--	0	--	0	--
W2	North Basin	38	0	--	0	--	0	--	0	--
Onondaga Lake, 1999										
W1	South Basin	1	0	--	0	--	0	--	0	--
W2	North Basin	1	0	--	0	--	0	--	0	--
W50	Willis Lakeshore Exposure Area	1	1	6.3	1	0.2	0	--	0	--
W51	Observed Fish Area	1	0	--	0	--	0	--	0	--
W52	Access from Fairgrounds	1	0	--	0	--	0	--	0	--
W53	Beach Access	2	0	--	0	--	1	0.5	0	--
W54	Lake Park Lakeland	1	0	--	0	--	0	--	0	--
W55	Harbor Brook	1	1	0.11	0	--	1	0.3	0	--
W56	Park/Picnic Area/Playground	1	0	--	0	--	0	--	0	--
W57	Boat Ramp (Liverpool)	1	0	--	0	--	0	--	0	--
W58	Lake Park Galeville	1	0	--	0	--	0	--	0	--

Source: Onondaga Lake Database
Notes: -- - not detected
Detailed information for each water sample is presented in Appendix B.

Table 8-3. Summary of Concentrations of Chlorinated Benzenes in Tributary and Onondaga Lake Water and Metro Discharge in 1992 and 1999

			Monochloro-benzene (µg/L)		Dichlorobenzenes (µg/L)						Trichlorobenzenes (µg/L)					
					1,2		1,3		1,4		1,2,3		1,2,4		1,3,5	
Station	Location	No. of Samples	No. Det.	Max. Conc.	No. Det.	Max. Conc.	No. Det.	Max. Conc.	No. Det.	Max. Conc.	No. Det.	Max. Conc.	No. Det.	Max. Conc.	No. Det.	Max. Conc.
Tributaries/Metro Outfall–Base Flow, 1992																
W3	Metro Outfall	3	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W4	Onondaga Creek	4	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W5	Harbor Brook	7	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W6	Ley Creek	3	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W7	East Flume	10	1	7.6	4.0	10	0	--	8	13	0	--	0	--	0	--
W8	Tributary 5A	7	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W9	Bloody Brook	1	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W10	Ninemile Creek	3	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W11	Sawmill Creek	1	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W12	Lake Outlet	2	0	--	0	--	0	--	0	--	0	--	0	--	0	--
Tributaries/Metro Outfall–Intermediate Flow, 1992																
W3	Metro Outfall															
W4	Onondaga Creek	7	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W5	Harbor Brook	3	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W6	Ley Creek	5	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W7	East Flume	5	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W8	Tributary 5A	3	0	--	0	--	0	--	2	2.3	0	--	2	1.1	0	--
W10	Ninemile Creek	6	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W12	Lake Outlet	7	0	--	0	--	0	--	0	--	0	--	0	--	0	--
Tributaries/Metro Outfall–High Flow, 1992																
W3	Metro Outfall	8	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W4	Onondaga Creek	5	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W5	Harbor Brook	9	0	--	1	1.0	0	--	2	7.2	0	--	0	--	0	--
W6	Ley Creek	4	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W7	East Flume	4	0	--	3	5.2	1	1.0	4	20	0	--	3	2.4	0	--
W8	Tributary 5A	9	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W9	Bloody Brook	1	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W10	Ninemile Creek	5	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W11	Sawmill Creek	1	0	--	0	--	0	--	0	--	0	--	0	--	0	--
W12	Lake Outlet	16	0	--	0	--	0	--	0	--	0	--	0	--	0	--
Onondaga Lake, 1992																
W1	South Basin	60	0	--	1	1.7	1	1.2	1	1.7	1	2.7	1	2.4	1	1.6
W2	North Basin	38	0	--	0	--	0	--	0	--	0	--	0	--	0	--
Onondaga Lake, 1999																
W1	South Basin	1	0	--	0	--	0	--	1	0.2	0	--	0	--	0	--
W2	North Basin	1	0	--	0	--	0	--	1	0.2	0	--	0	--	0	--
W50	Willis Lakeshore Exposure Area	1	1	12	1	3.2	0	--	1	3.4	0	--	0	--	0	--
W51	Observed Fish Area	1	0	--	1	0.1	0	--	1	0.3	0	--	0	--	0	--
W52	Access from Fairgrounds	1	0	--	0	--	0	--	1	0.2	0	--	0	--	0	--
W53	Beach Access	2	0	--	0	--	0	--	1	0.2	0	--	0	--	0	--
W54	Lake Park Lakeland	1	0	--	0	--	0	--	1	0.2	0	--	0	--	0	--
W55	Harbor Brook	1	1	0.5	1	0.2	0	--	1	0.2	0	--	0	--	0	--
W56	Park/Picnic Area/Playground	1	0	--	0	--	0	--	1	0.5	0	--	0	--	0	--
W57	Boat Ramp (Liverpool)	1	0	--	0	--	0	--	1	0.1	0	--	0	--	0	--
W58	Lake Park Galeville	1	0	--	0	--	0	--	0	--	0	--	0	--	0	--

Source: Onondaga Lake Database
Notes: -- - not detected
Detailed information for each water sample is presented in Appendix B.

Table 8-4. Summary of Honeywell Whole Fish and Fillet Data

	Data	Fillet	Whole Fish	Composite (YOY)	Composite (Adults)	Remains (Adults)	Total No. Samples	Ratio of Fillet to Whole Fish
Mercury (µg/kg)	No. of Analyses	11	22	15	100	11	159	
	No. of Detects	11	22	15	100	11		
	Mean	531	571	108	882	344		1.1
	Minimum	230	175	48	296	129		
	Maximum	904	1,400	224	1,890	734		
Total PCBs (µg/kg)	No. of Analyses	11	17	15	---	11	54	
	No. of Detects	11	17	15	---	11		
	Mean	686	1,718	619	---	1,467		2.5
	Minimum	88	300	142	---	380		
	Maximum	1,840	3,400	2,370	---	2,700		
DDT and metabolites (Sum) (µg/kg)	No. of Analyses	11	17	15	---	11	54	
	Mean	81	185	27	---	199		2.3
	Minimum	9.3	25	12	---	21		
	Maximum	208	499	87	---	443		
TEQ (1/2 DL) Avian (ng/kg)	No. of Analyses	11	18	16	---	11	56	
	No. of Detects	11	18	16	---	11		
	Mean	16	26	3.9	---	31		1.7
	Minimum	0.5	1.9	1.0	---	0.9		
	Maximum	101	128	13	---	194		
TEQ (1/2 DL) Mammalian (ng/kg)	No. of Analyses	11	18	16	---	11	56	
	No. of Detects	11	18	16	---	11		
	Mean	6.8	12	1.4	---	13		1.8
	Minimum	0.3	0.6	0.3	---	0.4		
	Maximum	41	62	4.7	---	79		

Notes:
1) Statistics are from samples collected in 1992 and 2000.
2) The data collected by PTI are included with the Honeywell samples.
3) 1992 Honeywell PCB data were excluded based on QA/QC.

Table 8-5. Uptake Factors Used to Estimate Prey Concentrations of COCs in Wildlife Receptor Diets

Analyte	Benthic Invertebrate Uptake Factors[a]	Earthworm Uptake Factors[b]	Small Mammal Uptake Factors[c]
Metals			
Arsenic	0.127	Not applicable	ln(mammal)= -4.5796+(0.7354*ln[soil])
Barium	ln(invert)=0.0395+(0.6092*ln(sed))[d]	0.091	Not applicable
Beryllium	Not applicable	0.045	Not applicable
Cadmium	9.24	ln(earthworm)=2.114+(0.795*ln(soil))	Not applicable
Chromium	log(invert)=0.2092+(0.365*log(sed))	Not available	ln(mammal)=-1.4945+(0.7326*ln(soil))
Cobalt	log(invert)=1.48+(-0.425*log(sed))[p]	Not applicable	Not applicable
Copper	log(invert)=1.037+(3.59*log(sed))	Not applicable	Not applicable
Lead	0.066	ln(earthworm)=-0.218+(0.807*ln(soil))	ln(mammal)=0.0761+(0.4422*ln(soil))
Manganese	0.066[f]	Not applicable	Not applicable
Mercury	e	ln(earthworm)=-0.684+(0.118*ln[soil])	0.0543
Methylmercury	e	ln(earthworm)=-0.684+(0.118*ln[soil])	0.192
Selenium	log(invert)=0.2092+(0.365*log(sed))[q]	ln(earthworm)=-0.075+(0.733*ln[soil])	Not applicable
Nickel	log(invert)=1.48+(-0.425*log(sed))	Not available	Not applicable
Thallium	0.066[f]	ln(earthworm)=-0.218+(0.807*ln(soil))[f]	Not applicable
Vanadium	0.127[g]	0.042	ln(mammal)=-4.5796+(0.7354*ln(soil))[g]
Zinc	log(invert)=1.77+(2.42*log(sed))	Not applicable	Not applicable
Volatile Organic Compounds			
Dichlorobenzenes	4.51[h]	Not applicable	Not applicable
Trichlorobenzenes	4.9	ln(earthworm)=3.533+(1.182*ln(soil))[i]	Not applicable
Xylenes	1	Not applicable	Not applicable
Semivolatile Organic Compounds			
Bis(2-ethylhexyl)phthalate	1	Not applicable	Not applicable
Hexachlorobenzene	4.17	ln(earthworm)=3.533+(1.182*ln(soil))[i]	(ln[mammal]= 0.8113+1.0993(ln[soil])[k]
Total PAHs	0.55[j]	ln(earthworm)=3.533+(1.182*ln(soil))[i]	(ln[mammal]= 0.8113+1.0993(ln[soil])[k]
Polychlorinated Biphenyls/Pesticides/Dioxins/Furans			
Chlordanes	Not applicable	ln(earthworm)=3.533+(1.182*ln(soil))[i]	Not applicable
DDT and metabolites	0.59[l]	Not applicable	(ln[mammal]= 0.8113+1.0993(ln[soil])[k]
Dieldrin	8.5	ln(earthworm)=3.533+(1.182*ln(soil))[i]	(ln[mammal]= 0.8113+1.0993(ln[soil])[k]
Total polychlorinated biphenyls	1.48	ln(earthworm)=1.410+(1.361*ln[soil])	(ln[mammal]= 0.8113+1.0993(ln[soil])[k]
Dioxins	0.45[m]	ln(earthworm)=3.533+(1.182*ln(soil))[i]	(ln[mammal]= 0.8113+1.0993(ln[soil])[n]
Furans	0.45[m]	ln(earthworm)=3.533+(1.182*ln(soil))[i]	0.1251[o]

Table 8-5. (cont.)

Notes:

Uptake factors are only provided for COCs in receptor prey (see footnotes a, b, and c below for receptors consuming each prey item).

[a] Benthic invertebrate uptake factors were based on USDOE (1998) for metals and USACE BSAF database for organics, except for mercury for which data were available. Receptors feeding on aquatic invertebrate prey include: mallard, tree swallow, little brown bat, mink, and river otter.

[b] Earthworm uptake factors were based on Sample et al. (1998a). When a UF was not available for a contaminant, professional judgment was used. The short-tailed shrew represents receptors feeding on terrestrial invertebrates.

[c] Small mammal uptake factors were based on Sample et al. (1998b). When a UF was not available for a contaminant, professional judgment was used. Receptors feeding on small mammals include the red-tailed hawk and mink.

[d] Cadmium regression equation applied for barium.

[e] Measured concentrations of mercury in invertebrates were used.

[f] Median lead uptake factors applied for manganese and thallium.

[g] Arsenic uptake factor applied for vanadium.

[h] Average of dichlorobenzene uptake factors used.

[i] 90th percentile earthworm UF for 2,3,7,8-TCDD (22.2) was applied to trichlorobenzenes, Hexachlorobenzene, PAHs, chlordanes, dieldrin, and all dioxins and furans.

[j] Average of PAH uptake factors used.

[k] 90th percentile small mammal UF for 2,3,7,8-TCDD (2.2) was applied to hexachlorobenzene, PAHs, DDT and metabolites, Dieldrin and PCBs.

[l] Average of DDT, DDE, and DDD uptake factors used.

[m] Average of dioxin and furan uptake factors used (both equal to 0.45).

[n] The recommendation for general estimates for tetrachlorodibenzo-p-dioxin was applied to all dioxins.

[o] The recommendation for general estimates for tetrachlorodibenzo-furan was applied to all furans.

[p] Nickel benthic uptake factor applied for cobalt.

[q] Chromium benthic uptake factor applied for selenium.

Table 8-6. Avian Receptor Life History Parameters

Factors	Units	Tree Swallow	Mallard Duck	Belted Kingfisher	Great Blue Heron	Osprey	Red-tailed Hawk
Body weight	grams	20.6 [a]	1,043 [g]	136 [k]	2,200 [l]	1,568 [o]	1,224 [l]
Food ingestion rate (dw basis)	kg/kg-day	0.264 [b]	0.101 [b]	0.137 [b]	0.045 [b]	0.048 [b]	0.052 [b]
Water ingestion rate	kg/kg-day	0.212 [c]	0.058 [c]	0.114 [c]	0.045 [c]	0.051 [c]	0.055 [c]
Percent Diet Composition							
Fish 3 to 18 cm				100%	33%	10%	
Fish greater than 18 cm					67%	90%	
Aquatic Invertebrates		100%	50%				
Aquatic Plants			50%				
Vertebrate prey (e.g., mammals, birds)							100%
Sediment ingestion rate (dw basis)	% of FIR	0% [d]	3.3% [h]	1% [d]	1% [d]	0% [d,j]	1% [d,j]
Foraging radius (mean/maximum)	km	0.1/0.2 [e]	111/540 [i] hectares	0.4 -1.0 [k]	3.1/24 [m]	1.7 to 10 [p] (mean)	233/1770 [p,q] hectares
Residence time (maximum)	days/yr	365 [f]	365 [f,j]	365 [f]	365 [f,n]	243 - 365 [d]	365 [f]

Notes: -- - assumed to forage onsite exclusively
FIR - food ingestion rate
NA - not available
SD - standard deviation

[a] Secord and McCarty (1997).
[b] Based on Nagy (1987).
[c] Based on Calder and Braun (1983).
[d] Based on professional judgement.
[e] McCarty and Winkler (1999).
[f] Cornell University (2001).
[g] Nelson and Martin (1953).
[h] Estimated from Beyer et al. (1994).
[i] Dywer et al. (19879), Kirby et al. (1985).
[j] Andrle and Carroll (1988).
[k] Brooks and Davis (1987). Non-breeding range.
[l] Dunning (1993).
[m] Dowd and Flake (1985).
[n] Bull (1998)
[o] Brown and Amadon (1968).
[p] USEPA (1993b).
[q] Janes (1984).

Table 8-7. Mammalian Receptor Life History Parameters

Factors	Units	Little Brown Bat	Short-tailed Shrew	Mink	River Otter
Body weight	grams	7.1 [a]	15 [f]	600 [k]	5,450 [n]
Food ingestion rate (dw basis)	kg/kg-day	0.102 [b]	0.157 [d,g]	0.0643 [l]	0.044 [l]
Percent Diet Composition					
Fish 3 to 18 cm				35%	30%
Fish greater than 18 cm					60%
Aquatic Invertebrates		100%		15%	10%
Vertebrate prey (e.g., mammals, birds)				50%	
Terrestrial Invertebrates			100%		
Water ingestion rate	kg/kg-day	0.162 [c]	0.151 [c]	0.104 [c]	0.084 [c]
Sediment ingestion rate (dw basis)	% of FIR	0% [d]	13% [h]	1% [d]	1% [d]
Foraging radius (mean/maximum)	km	0.1 [e]	0.05/0.22 [i] hectares	1.85/2.8 [m]	9/22.5 [o]
Assumed Residence time	days/yr	365 [d]	365 [j]	365 [n]	365 [d]

Notes: - assumed to forage onsite exclusively
FIR - food ingestion rate
NA - not available
SD - standard deviation

[a] Bopp (1999).

[b] Hoffman, (1999); Environment Canada (2000); Synder (2002).

[c] Based on Calder and Braun (1983).

[d] Based on professional judgement.

[e] Buchler (1976).

[f] Schlesinger and Potter (1974).

[g] Schmidt (1994).

[h] Talmage and Walton (1993).

[i] Platt (1976).

[j] Nowak (1997).

[k] Mitchell (1961).

[l] Based on Nagy (1987).

[m] Gerell (1970).

[n] Allen (1986).

[o] Spinola et al. (undated).

9. ANALYSIS OF ECOLOGICAL EFFECTS (ERAGS STEP 6)

In accordance with USEPA guidance, this chapter describes information on observed effects for ecological components of the Onondaga Lake ecosystem. The groups of ecological receptors discussed in Section 9.1 are aquatic macrophytes, phytoplankton, zooplankton, fish, herpetofauna (amphibians and reptiles), and wildlife (birds and mammals). Most of the information on these groups was collected by other parties and is presented in detail in the publications cited herein.

Benthic invertebrates are discussed in detail in Section 9.2, which contains a detailed evaluation of sediment toxicity and benthic macroinvertebrate communities. Most of the data were collected by Honeywell in 1992 and 2000 as part of the remedial investigation (RI). This section also contains a key element of the effects characterization, the development of site-specific sediment effect concentrations (SECs) for Onondaga Lake, using the empirical information collected on sediment chemistry and sediment toxicity during the RI. The development of site-specific SECs is consistent with the recommendations of NYSDEC (1994b, 1997a) to conduct site-specific evaluations based on sediment toxicity tests when chemical concentrations in sediments are found to exceed the state sediment screening values.

The information on benthic macroinvertebrate communities was not used to develop site-specific SECs, because it was found that benthic communities at every station in the lake are impaired to some degree; thus, it is not possible to calculate an SEC because these calculations require that a certain proportion of stations have no effects. Although this lakewide impairment is typical of eutrophic lakes, it can reduce the value of these communities for developing site-specific SECs, particularly if there is ambiguity as to the identity of the factors primarily responsible for any observed community effects. The results of the benthic macroinvertebrate evaluations for Onondaga Lake were, therefore, used primarily to interpret the magnitude and significance of any potential sediment toxicity predicted on the basis of the site-specific SECs.

The last section of this chapter, Section 9.3, presents the effects characterization for aquatic and terrestrial vertebrates. It describes the methodology used and the toxicity reference values (TRVs) selected for fish, avian, and mammalian receptors.

9.1 Onondaga Lake Field Studies/Observations

9.1.1 Aquatic Macrophytes

Aquatic macrophytes form an important part of lake ecosystems. They serve as food for other aquatic organisms and provide habitat for insects, fish, and other aquatic and semi-aquatic organisms. Most aquatic macrophytes are rooted or attached to the sediment, although some free-floating forms exist (Auer et al., 1996a). Little quantitative information existed on macrophyte distributions in Onondaga Lake prior to 1991, when Madsen et al. (1993) conducted the first quantitative survey of macrophyte distributions in the lake, as discussed in Chapter 3, Section 3.2.5.1.

In addition to conducting a field survey of lake macrophytes, Madsen et al. (1993) collected different kinds of sediments from various parts of the littoral zone and used them for greenhouse experiments in which macrophyte growth was evaluated in each sediment type. Two additional field surveys of macrophytes in the lake were conducted by Honeywell in 1992 and 1995 as part of the RI.

During the 1991 field survey conducted by Madsen et al. (1993), five species of submerged macrophytes species were found in Onondaga Lake:

- *Ceratophyllum demersum* (coontail).
- *Heteranthera dubia* (water stargrass).
- *Myriophyllum spicatum* (Eurasian water-milfoil).
- *Potamogeton crispus* (curly-leaf pondweed).
- *P. pectinatus* (Sago pondweed).

An additional five aquatic macrophyte species were found by Madsen et al. (1998) in 1993. These include *Potamogeton diversifolius* (waterthread pondweed), *Elodea canadensis* (Canadian pondweed), *Lemna minor* (duckweed), *Sparganium* sp. (bur-reed), and *Zannichellia palustris* (horned pondweed). All of the newly reported species were relatively rare, with the possible exception of *Elodea canadensis*. The distribution of these species throughout the littoral zone of the lake was found to be relatively limited. The number of macrophyte species found in the lake (i.e., ten) is low, since up to 15 were present in Onondaga Lake before 1940 (Auer et al., 1996a). For a eutrophic lake in New York State, 15 species of vascular plants is typical (Madsen et al., 1993), which is lower than the New York State average of 18 (Madsen et al., 1996).

Madsen et al. (1993, 1996) also conducted laboratory studies to evaluate the role of sediments in limiting plant growth. They planted *P. pectinatus,* the dominant aquatic plant in Onondaga Lake, in lake sediments collected from nine locations that corresponded to areas of high, medium, and low macrophyte abundance, as well as four different sediment types (oncolite, organic, sand, and silt). In addition, the authors also collected a silty reference sediment from a lake in Texas. Madsen et al. (1993, 1996) found that growth on the fertile reference sediment was significantly higher than growth on Onondaga Lake sediments. They predicted that improvement in water clarity or quality alone would not improve plant growth, as sediment degradation is directly related to the input of $CaCl_2$ (calcium chloride) into the lake and resulting calcium carbonate deposition. Sediments from sites with high, medium, and low percentages of plant cover showed significant differences in their abilities to support plant growth, with laboratory results corresponding to plant cover seen in the field.

The work of Madsen et al. corresponds with the macrophyte transplant study performed in the summer of 1992 by Honeywell (PTI, 1993c). The purpose of this study was to determine the extent to which representative macrophyte species can survive and grow in the sediment and water of the littoral zone of Onondaga Lake (PTI, 1993c). Three macrophyte species (*P. pectinatus, P. richardsonii* [redhead grass], and *Vallisneria americana* [wild celery]) were transplanted at two depths (1 and 1.5 meters [m]) at six locations in the littoral zone of Onondaga Lake (Figure 9-1) and at two potential reference lakes (Otisco and Cross Lakes, of which only the former was determined to be an appropriate reference lake and is discussed here). A total of three seeded racks (one for each macrophyte) were deployed at each location at the two depths in Onondaga Lake. Each rack contained three sediment samples from each of the reference lakes and four from Onondaga lake for a total of 10 samples per rack and 30 samples per depth at each location. Twelve samples (three plant species × four reference lake sediment samples) were deployed at two depths at one location in the reference lakes.

The results, as summarized in the table below, showed macrophyte survival to be minimal at Onondaga Lake. The lakewide survival rate was less than 3 percent at 1 m, and improved to slightly under 12 percent at 1.5 m. In contrast, 75 and 58 percent survival rates were seen at depths of 1 and 1.5 m at Otisco Lake.

Macrophyte Transplant Study Results (PTI, 1993c)

Location	Mean Percent Survival		No. of Plants Surviving	
	1 m	1.5 m	1 m	1.5 m
Onondaga Lake M1 (between Harbor Brook and Onondaga Creek)	0	0	0/30	0/30
Onondaga Lake M2 (between Onondaga Creek and Ley Creek)	6.7	6.7	2/30	2/30
Onondaga Lake M3 (south of Tributary 5A)	0	27	0/30	8/30
Onondaga Lake M4 (mouth of Ninemile Creek)	6.7	0	2/30	0/30
Onondaga Lake M5 (northwest corner of lake)	3.3	13	1/30	4/30
Onondaga Lake M6 (south of Sawmill Creek)	0	23	0/30	7/30
Otisco Lake	75	58	9/12	7/12

Based on these results, it appears that stressors limit macrophyte growth in Onondaga Lake. According to Auer et al. (1996a), the major stressors potentially limiting macrophytes in Onondaga Lake include the following:

- **Water transparency:** Reduced water transparency in the water column is a major factor in limiting the depth distribution of submerged macrophytes (Canfield et al., 1985; Chambers and Kalff, 1985). Historically, transparency in Onondaga Lake has been very low (<1 m), primarily because of the eutrophic nature of the lake (Effler and Perkins, 1996b). More recently, however, water clarity in the lake has been improving (Effler and Perkins, 1996a), increasing the amount of time in which a plant can complete its life cycle before parent-plant mortality (Auer et al., 1996a).

- **Salinity:** The high levels of salinity that have prevailed in Onondaga Lake in the past may have prohibited the establishment of many common emergent and floating-leaved species, due to increased stress from evapotranspiration (Auer et al., 1996a). Although the salinity of the lake has declined (Effler et al., 1996) and is now within the tolerance range of more species, the dominance of *P. pectinatus* in many habitats may hinder the colonization of returning species.

- **Calcium carbonate precipitation:** The extremely high rate of calcite ($CaCO_3$) precipitation and deposition onto the surfaces of macrophytes in Onondaga Lake in the past may have been sufficient to completely coat plants (Auer et al., 1996a). This mechanism may have been responsible for or contributed to the disappearance of charophytes from Onondaga Lake (Dean and Eggleston, 1984). Calcium carbonate deposition decreased 64 percent from 1985 to 1989, reflecting the relative decrease in external loading (70 percent) after the closure of the soda ash/chlor-alkali facility in 1986 (Effler et al., 1996). However, high concentrations of calcite are still present in and being released to the lake (Effler et al., 1996).

- **Oncolites:** During the 1991 macrophyte survey conducted by Madsen et al. (1993), macrophyte distributions were found to be limited in areas where oncolites were present. The greenhouse experiments conducted by Madsen et al. (1993) indicated that macrophytes grow as well on oncolite sediment as on the other kinds of sediment in the lake. Madsen et al. (1996a) suggested that oncolites may limit macrophytes in the field because their relatively low density makes them susceptible to movement by wave action; therefore, oncolites may provide an unstable substrate for macrophyte colonization.

- **Water level changes:** As described in Chapter 3, the level of Onondaga Lake is regulated as part of the New York State Barge Canal System. Auer et al. (1996a) noted that although annual variations in lake level are usually less than 1 m, even that magnitude of change can have a substantial effect on macrophytes since light restricts macrophyte growth to shallow depths.

Based on the information presented by Auer et al. (1996a), it appears that many potential stressors have become less limiting to macrophytes in Onondaga Lake in recent years. Recently, Madsen et al. (1998) reported on a series of experiments in Onondaga Lake to evaluate techniques for restoration of littoral areas to improve fish habitat. Two temporary habitat areas were created with wave breaks (i.e., hay bales) and fencing for protection of natural colonization by macrophytes, as well as planted macrophytes. The authors found that survival of most planted species was low. Potential causes of low survival include herbivory (primarily by waterfowl) and changes in water levels by 0.3 m after planting was performed. However, the habitats were successful in enhancing colonization by native macrophyte species.

Between July and September, the authors found large amounts of filamentous algae in the habitats, which is generally detrimental to the growth of rooted macrophytes (i.e., through light limitation, competition for nutrients, and mechanical damage). Madsen et al. (1998) found mechanical damage for several emergent species, where floating masses of filamentous algae, driven by wind and wave action, collapsed emergent macrophyte stems and leaves.

9.1.2 Phytoplankton

Phytoplankton communities have been routinely monitored at two stations in Onondaga Lake since 1970 by Onondaga County, following a detailed study of lake conditions in 1969. This monitoring involves collection of water samples, usually biweekly, over the spring to early fall interval from the south deep station (W1) at the surface and at depths of 3, 6, and 12 m (Auer et al., 1996a). This information has been summarized and interpreted by Auer et al. (1996a).

In general, the characteristics of the phytoplankton communities of Onondaga Lake have reflected the eutrophic nature of the lake. Prior to 1972 (when phosphorus loadings were dramatically reduced due to a local ban on phosphates in detergents), cyanobacteria (formerly known as blue-green algae) were common in the lake during the spring-fall growing season. A seasonal succession was described in which the major groups were diatoms in spring, green algae in early summer, and cyanobacteria in the late summer and fall (Auer et al., 1996a). Although cyanobacteria disappeared from the lake after 1972, it returned in the late 1980s, apparently due to more efficient grazing zooplankton, as cyanobacteria may not be a suitable food for zooplankton (Auer et al., 1996a). No obvious changes were observed in nutrient loading during that period. Auer et al. (1996a) noted that as the degree of eutrophy in the lake has declined since the early 1970s, the intensity of phytoplankton blooms has declined. However, the authors also noted that Onondaga Lake remains highly eutrophic and that concentrations of phosphorus in the lake remain sufficient to sustain near-maximum rates of algal growth over the entire summer. The strong seasonal changes in phytoplankton biomass seen in Onondaga Lake represent imbalances between growth and loss processes.

Based on the information summarized by Auer et al. (1996a), the primary stressors that have affected phytoplankton communities in Onondaga Lake are nutrients, which have influenced both the types of species found in the lake and the densities of those species. Although the effect of mercury contamination on the phytoplankton community is unknown, it is evident from the bioaccumulation investigation (PTI, 1993b) that mercury accumulates in phytoplankton and can be passed on to animals feeding on phytoplankton in Onondaga Lake.

9.1.3 Zooplankton

Zooplankton communities in Onondaga Lake have been routinely monitored by Onondaga County in conjunction with the phytoplankton monitoring. This information has been summarized and interpreted by Siegfried et al. (Auer et al., 1996a; Siegfried et al., 1996).

The number of zooplankton species found in Onondaga Lake has increased substantially since the early 1970s. This increase can be attributed to both the increased sampling effort, which has collected a number of rare species, and the closure of the Honeywell soda ash/chlor-alkali facilities in 1986 (Auer et al., 1996a; Siegfried et al., 1996). The soda ash/chlor-alkali process was in operation from 1884 to 1986 and released large quantities of ionic waste (high in calcium, chloride, and sodium ions) into Onondaga Lake, impacting the ecology so that the salinity of the lake and the rate of calcium carbonate precipitation were artificially high (Siegfried et al., 1996).

Salinity affects the osmoregulation capabilities of zooplankton, while calcium carbonate particles can physically interfere with feeding. The chloride/salinity levels of the lake before the closure of the facility were near the upper limit for freshwater organisms. Since the facility has been closed, the salinity and transparency (clear water phases) of the lake have improved. The relative abundance of species within the zooplankton community has also changed considerably since the 1970s. Prior to the mid-1980s, the zooplankton biomass was dominated by *Cyclops vernalis*, a cyclopoid copepod. Since that time, however, dominance has shifted to large-bodied cladocerans (*Daphnia* spp.) and the calanoid copepod *Diaptomus siciloides*. During the period of peak pollution in the 1970s and 1980s native species of Daphnia were replaced by exotic species, such as *Daphnia exilis* and *D. curvirostris*. These exotic species are found in saline environments in the southwestern US and Europe (Hairston et al., 1999, Duffy et al., 2000).

Mercury has also affected the Onondaga Lake zooplankton community. The period of peak mercury concentrations in the sediments based on ^{210}Pb dating, coincides with zero hatching success of *D. exilis* eggs in laboratory monitoring (Hairston et al., 1999). Whether mercury in the water column caused the eggs to become non-viable at the time they were produced, or mercury and/or other contaminants and stressors in the sediments made the eggs non-viable over the burial period, is uncertain.

Despite recent increases in zooplankton diversity, the zooplankton assemblage of the lake remains depauperate compared to other lakes in the region (Auer et al., 1996a). Further reductions in the loadings of ionic waste-associated stressors may result in additional changes to the zooplankton community.

9.1.4 Fish

Most quantitative studies of fish communities in Onondaga Lake have been conducted by researchers at the State University of New York College of Environmental Science and Forestry (SUNY ESF) since 1989, and results of those studies have been summarized by Ringler et al. (Auer et al., 1996a). Only qualitative information exists on fish communities in the lake prior to 1989.

A total of 55 fish species have been collected in Onondaga Lake since 1989 (see Table 3-7). Although this number represents a considerable increase over the numbers found in historical studies, comparison of current species richness with historical values is difficult because considerably more sampling effort was used in the more recent studies, increasing the probability of collecting more species (Auer et al., 1996a; Tango and Ringler 1996). The current level of species diversity in Onondaga Lake is similar to values found in other New York lakes, and growth rates, age distributions, and mortality rates of several species are similar to those observed in other northeastern US lakes (Auer et al., 1996a).

However, in contrast to comparison lakes, many of the species found in Onondaga Lake do not reproduce there and recruitment rates are unknown. Only 16 of 48 species captured in 1991 were found to reproduce in the lake, and reproduction within the lake varied by location. Based on the absence of juveniles in the catches of shoreline seine hauls, species such as the walleye and northern pike are thought not to reproduce in the lake. Areas characterized by the presence of aquatic macrophytes and submerged structures (e.g., near the lake outlet) supported the largest populations of juveniles. Areas with heavy silt loads and that are unprotected from wind are undesirable as spawning areas, as silt loads or wave action may cause eggs to be covered or removed from optimal areas (Auer et al., 1996a).

As discussed in Section 3.2.5.1, Onondaga Lake supports a warmwater fish community that is dominated by the pollution-tolerant gizzard shad (*Dorosoma cepedianum*), freshwater drum (*Aplodinotus grunniens*), carp (*Cyprinus carpio*), and white perch (*Morone americana*) (Auer et al., 1996a). Food-habit studies documented important prey species in the lake, including fish, fish eggs, zooplankton (primarily copepods and cladocerans), and benthic macroinvertebrates (primarily chironomids). However, the absence of macrophytes from many areas of the lake reduces the use of those areas by fish.

The composition of the fish community in the lake varies seasonally, with migration between the Seneca River and the lake being an important contributor to the variability. One of the major changes in the fish community occurs during fall turnover, when concentrations of dissolved oxygen decline throughout the water column. Based on reduced catches conducted during fall turnover and a complimentary increase in Seneca River catches, it is likely that many fish leave the lake to avoid the stress of low dissolved oxygen concentrations (Auer et al., 1996a). Species moving out of the lake include channel catfish (*Ictalurus punctatus*), gizzard shad, white perch, and smallmouth bass (*Micropterus salmoides*) (Auer et al., 1996a).

According to Ringler et al. (Auer et al., 1996a), the major stressors that are potentially limiting to fish in Onondaga Lake include the following:

- **Dissolved oxygen** – The absence of dissolved oxygen concentrations in the hypolimnion of the lake during stratification and the unusually low concentrations throughout the water column during fall turnover represent significant constraints on fish in the lake.

- **Absence of macrophytes in parts of the lake** – Macrophytes play a critical role in supporting the fish community of Onondaga Lake, as demonstrated by the absence of juveniles and adults in areas of low macrophyte density.

Ringler et al. (1995) evaluated the nesting activity of fish in the nearshore zone of Onondaga Lake, and found that the highest densities of nests were located on the northwestern shoreline between the lake outlet and the mouth of Ninemile Creek. They noted that this shoreline is composed primarily of oncolytic sediments and is relatively well protected from the predominantly northwest winds. Ringler et al. (1995) stated that conclusions regarding relationships between nest densities and habitat variables cannot be drawn without more quantitative data on distribution and abundance of habitat types. From a qualitative standpoint, however, they noted that the flat, unvegetated habitat off the wastebeds on the western shoreline of the lake provides minimal cover for spawning fish, and relatively few nests were found in that area. Ringler et al. (1995) also noted that the paucity of nests in the entire southern part of the lake may be related to the sparse distribution of macrophytes in certain areas, turbidity from Onondaga Creek, or ammonia mainly from the Metropolitan Syracuse Sewage Treatment Plant (Metro) effluent.

9.1.5 Amphibians and Reptiles

Between April and October of 1994, Ducey and Newman (1995) conducted a herpetological survey along the perimeter of Onondaga Lake. They made 30 visits to the lake and expended approximately 235 person-hours during the surveys. This study is the first and only comprehensive assessment of herpetofauna near the lake.

Ducey and Newman (1995) documented the presence of seven amphibian species (i.e., five frog and two salamander species) and six reptilian species (i.e., three snake and three turtle species) near Onondaga Lake during their 1994 survey. They found that habitats around the lake differed dramatically in the species supported, with the lake itself and many other areas nearly devoid of herpetofauna. The terrestrial areas around the lake were divided into five regions for evaluation. The results were as follows:

- **Region A:** This region is located along the northwest shoreline of Onondaga Lake, and includes Maple Bay, Ninemile Creek, several wooded areas, a large swamp, many temporary wetlands, two ponds (one connected to the lake and the other isolated from the lake), and several fields with grass and shrubs. According to Ducey and Newman (1995), this region supported, by far, the largest numbers of individuals and the greatest number of species. However, even within this region, herpetofaunal populations varied. No herpetofauna were found at the pond with hydrological connections to the lake.

- **Region B:** This region is located along the northeast shoreline of Onondaga Lake, and includes a park area along Willow Bay, a drainage ditch, a seasonally flooded forest, Sawmill Creek, and a dump area for the Parks Department. According to Ducey and Newman (1995), this region supported few amphibians and reptiles, despite extensive searching by the authors in spring, summer, and fall. The authors were particularly surprised by the fact that no frogs were ever heard calling, even during times when they were very active elsewhere. The authors also reported that no tadpoles or frogs were found within the forest or along Sawmill Creek, despite apparently ample water, vegetation, and insects. They also noted that no snakes were found in the dump area, despite the fact that it contained many kinds of suitable debris for habitat. Ducey and Newman (1995) recommended that Region B be reexamined to confirm the absence of herpetofauna in that area.

- **Region C:** This region is located along the southeast shoreline of Onondaga Lake between Onondaga Creek and Bloody Brook, and includes grass fields and a series of ponds (connected to the lake). Ducey and Newman (1995) found only a single turtle in this region, but speculated that the region probably supports one or two species of snakes. The authors concluded that they would not expect to find large numbers of herpetofauna in this region because of the lack of suitable cover.

- **Region D:** This region is located along the southwest shoreline of Onondaga Lake between Onondaga Creek and Tributary 5A, and includes extensive stands of *Phragmites australis* (common reed) along the shoreline, small forested areas, and broad grassy areas. Ducey and Newman (1995) noted that they did not investigate this region as extensively as Regions A, B, and C. They found moderate numbers of snakes, but no frogs or salamanders.

- **Region E:** This region is located along the western shoreline of Onondaga Lake between Tributary 5A and Ninemile Creek, and includes elevated Solvay wastebeds (with cliffs) and parking lots for the fairgrounds in the upper areas. Ducey and Newman (1995) found no herpetofauna in the elevated areas, but a small snake population was found on the lake shoreline. They concluded that the soil layers appeared to be too thin throughout most of this region to support adequate invertebrate prey populations or to provide ample subterranean tunnels for herpetofauna.

Ducey and Newman (1995) compared the results of their 1994 survey around Onondaga Lake with previous results from elsewhere in central New York State. They concluded that the herpetofauna around the lake was generally depauperate, and were surprised by the absence of some common species. They found that the seven amphibian and six reptilian species found around the lake were considerably fewer than the 19 amphibian and 15 reptilian species recorded for Onondaga County as a whole during 1990 to 1996 by NYSDEC (1997b).

Ducey (1997) conducted additional evaluations of the herpetofauna near Onondaga Lake between March 1995 and May 1997. For these studies, the author focused only on Region A along the northwest shoreline of the lake. The results confirmed that the Onondaga Lake littoral zone and shoreline supports no amphibian reproduction, in contrast to other moderately sized lakes in central New York. Five species of anurans and two species of salamanders utilized wetlands (not connected to the lake) and terrestrial habitats within 100 m of the lake. Red-spotted newts (*Notophthalmus viridescens*) were found in one of the unconnected wetlands at a density up to three orders-of-magnitude lower than found at other sites in central New York (Ducey, 1997).

Environmental factors that may be affecting herpetofaunal distribution include:

- High concentrations of ionic waste (chloride, sodium, and calcium ions) that may affect the physiological processes.
- Chemical contaminants.
- Effluent from the Metro sewage treatment plant.
- Poor habitat on the southern shores of the lake (Ducey and Newman, 1995).

Ducey et al. (2000) directly assessed the toxicity of water from the lake and wetlands on developing amphibian embryos. They found that water from connected wetlands and the lake has variable, but consistently negative, effects on amphibian development relative to controls. They hypothesized that there is a chemical interaction that affects amphibian embryos, because unfiltered Onondaga Lake water is highly toxic to embryos, while filtered water is not as toxic.

9.1.6 Terrestrial Plants

Various plant communities are found around Onondaga Lake (i.e., terrestrial, wetland, aquatic, or urban systems). The area around Onondaga Lake has been extensively modified, through development, for more than a century. The vegetation found on the wastebeds (Chapter 3) has been affected by activities at Honeywell facilities (i.e., disposal of Solvay wastes). Table A-1 of Appendix A lists characteristic flora of the ecological communities around Onondaga Lake.

9.1.7 Birds and Mammals

Tables 3-11 to 3-13 provide summaries of bird species observed around Onondaga Lake and Table 3-14 lists mammals potentially found in the area. Since there are no data available that could be used to correlate observed wildlife populations with contamination in the area, contaminant doses (estimated using food-web modeling) compared to TRVs (see Section 9.3) are the main measurement endpoint used to evaluate avian and mammalian assessment endpoints.

9.2 Benthic Macroinvertebrates/Sediment Effect Concentrations

The potential ecological risks posed to benthic macroinvertebrate communities by surface sediment in Onondaga Lake were evaluated using three kinds of information collected by Honeywell in 1992 and 2000 during the RI field investigation:

- Chemical concentrations in surface sediments.

- Sediment toxicity tests using the 10-day (1992) and 42/40-day (2000) tests, based on the amphipod *Hyalella azteca* and the midge *Chironomus tentans*.

- Evaluations of in situ benthic macroinvertebrate communities.

The three indicators used to evaluate surface sediment in Onondaga Lake (i.e., sediment chemical concentrations, sediment toxicity tests, and benthic community evaluations) are used to provide a weight-of-evidence approach regarding the risk of toxicity posed by sediment throughout the lake in Chapter 10. That is, the independent information provided by each of the three indicators will be combined to provide an integrated assessment of sediment quality that would not be possible using any single indicator (USEPA, 1997b). For example, sediment chemical concentrations provide information on which chemicals are elevated above sediment effect concentrations (SECs) or consensus probable effect concentrations (PECs) and may pose an ecological risk, but they do not provide the information to conclusively determine if those chemicals are sufficiently bioavailable to pose a risk. The sediment toxicity tests provide information on whether particular sediments are toxic to the test organisms, but they do not identify the toxic components of the sediment or determine whether the observed toxicity is sufficient to result in adverse effects to the resident benthic macroinvertebrate communities. Finally, alterations of benthic communities provide information on whether sediment may be adversely affecting resident organisms, but they do not identify the causative agents. The use of a weight-of-evidence approach to evaluate potential risks to benthic macroinvertebrates is consistent with USEPA guidance and recommendations (USEPA, 1997a,b).

The 114 stations sampled in Onondaga Lake and the eight tributary stations sampled around the perimeter of the lake for sediment evaluations in 1992 are presented on Figure 7-2 in Chapter 7. Sixty-six lake stations were sampled for sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities. Thirteen lake stations were sampled only for sediment toxicity and sediment chemistry. Thirty-five lake stations were sampled only for sediment chemistry. The eight tributary stations were sampled only for benthic macroinvertebrate communities. The 15 stations sampled in Onondaga Lake for sediment evaluations in 2000 are presented on Figure 7-6 in Chapter 7.

In conducting the sediment evaluations, Otisco Lake was used as the reference lake for determining the significance of sediment toxicity and effects on benthic macroinvertebrate communities at stations in Onondaga Lake. In addition, NYSDEC suggested in May 1999 that stations deeper than 3 m may be depth-impacted (Larson, pers. comm., 1999b). However, in order to provide a greater areal coverage of the lake bottom, the evaluation is limited to the 4.5 m contour for the 1992 benthic data and the 5 m contour for the 2000 benthic data.

In the following sections, results of the sediment evaluations are described, and site-specific SECs and PECs are developed using the empirical information collected on chemical concentrations and sediment toxicity in Onondaga Lake in 1992 and 2000. The site-specific PECs are used in Chapter 10 for risk characterization.

9.2.1 Results of Sediment Toxicity Tests

Results of sediment toxicity tests for each station sampled in Onondaga Lake in 1992 and 2000 and the reference lake (i.e., Otisco Lake) are presented in Tables 9-1 and 9-2, respectively.

Toxicity results from each station in Onondaga Lake were compared statistically with a single reference station in Otisco Lake. Separate evaluations were conducted for the data collected in 1992 and 2000. Paired comparisons between results for each station in Onondaga Lake and the results for Otisco Lake were made using the Washington State Department of Ecology SEDQUAL program (WSDE, 2001).

SEDQUAL performs statistical comparisons among test, reference, and control stations to identify stations exhibiting adverse effects. In SEDQUAL, test data may be compared to either reference data or control data. As stated above, in this BERA, Onondaga Lake stations were compared to single reference stations in Otisco Lake. Records are distinguished as reference/control data or test data by the sample use code. Statistical and data analysis features in SEDQUAL include:

- Wilks-Shapiro test for normality.
- Levene's test for equality of variances.
- Student's t-statistic, approximate t-statistic, Mann-Whitney, and rankits.
- User-specified reference station when a survey has more than one reference.
- Comparison of reference or control data to numeric performance standards.
- Optional use of negative control instead of reference if a survey has no reference stations or reference stations fail to meet a performance standard.

9.2.1.1 1992 Sediment Toxicity Results

Survival for the 10-day amphipod and chironomid toxicity tests was relatively high (i.e., >80 percent) at most stations in Onondaga Lake, indicating that lethal toxicity was not widespread throughout the lake (Figure 9-2). The chironomid test exhibited a greater range of survival than did the amphipod test.

Biomass also exhibited a greater range for the chironomid test than for the amphipod test (Figure 9-2). Amphipod biomass was lower than control values (i.e., <100 percent) at approximately half of the stations (47 percent) in Onondaga Lake, whereas chironomid biomass was less than control values at approximately one-third of the stations.

The greater variation in the responses of the chironomid test compared to the amphipod test may be due to the fact that the chironomids generally live in more direct contact with sediments than do the amphipods. While the amphipods live primarily on the sediment surface, the chironomids burrow into the sediments where they reside in cases (American Society for Testing and Materials [ASTM], 1993).

Values of mean survival for the five stations sampled in Otisco Lake were high for both the amphipod and chironomid tests (90 and 97 percent, respectively). Values of mean biomass for the amphipod and chironomid tests for the five stations sampled in Otisco Lake were similar to or greater than negative control values (i.e., 97 and 182 percent of control values, respectively).

Results of the statistical comparisons of the sediment toxicity data collected in 1992 are presented in Table 9-1. In general, the chironomid test was found to be a more sensitive indicator of sediment toxicity than the amphipod test. For example, statistically significant ($P \leq 0.05$) reductions of amphipod survival and growth were found at 1 and 18 stations, respectively, whereas significant ($P \leq 0.05$) reductions of chironomid survival and growth were found at 35 and 15 stations, respectively. The different sensitivities of the two test species may be related to the life-history patterns described above, where the amphipods live primarily on the sediment surface, while the chironomids burrow more deeply into the sediments.

For the 79 stations sampled in Onondaga Lake in 1992, effects based on amphipod survival, amphipod biomass, chironomid survival, and chironomid biomass were identified at 1, 18, 35, and 15 stations, respectively. If the results of all four toxicity endpoints are combined, they jointly identify effects at 40 stations. Of these 40 stations, the chironomid test independently identifies effects at most (i.e., 36 of 40, or 90 percent) of the stations at which any kind of toxicity was found.

The spatial patterns of amphipod and chironomid toxicity are presented in Figure 9-3. Most amphipod toxicity was confined to a small area in the southwestern corner of the lake, along Wastebeds 1 through 8 and along the Honeywell lakeshore area near Harbor Brook and the East Flume.

Most chironomid toxicity was confined to the southern half of the lake (Figure 9-3), although toxicity was also found in two areas in the northern half of the lake (i.e., off Ninemile Creek and near Sawmill Creek). In the southern half of the lake, lethal chironomid toxicity was found in three general areas: off Tributary 5A, off Ley Creek, and in the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume).

9.2.1.2 2000 Sediment Toxicity Results

The results of the statistical comparisons of the 42-day sediment toxicity data collected in 2000 are presented in Table 9-2. The spatial patterns of amphipod and chironomid toxicity are presented in Figure 9-4. In general, the patterns of toxicity observed for both tests were similar.

For the amphipod test, lethal toxicity was found at six stations (Figure 9-4), including all of the shallow (i.e., <5 m water depth) nearshore stations from Tributary 5A to the East Flume (Stations S332, S337, S342, S344, and S365) and near the Metro outfall (Station S317). Mean survival at those six stations ranged from 9 to 59 percent, compared to a mean value of 88 percent found for Otisco Lake. Amphipod biomass was found to be impacted at Stations S317 and S323, while reproduction was affected at three locations, Stations S342, S344, and S365.

For the chironomid test, lethal toxicity was found at nine stations (Figure 9-4), including all five of the shallow (i.e., <5 m water depth) nearshore stations from Tributary 5A to the East Flume (i.e., Stations S332, S337, S342, S344, and S365), two stations off Ninemile Creek (Stations S302 and S303), as well as the stations off Ley Creek (Stations S320 and S323). Mean survival at those nine stations ranged from 0 to 46 percent, compared to a mean value of 83 percent found for Otisco Lake. In addition to the nine stations at which lethal toxicity was found for the chironomid test, sublethal toxicity was found at Station S317 off Onondaga Creek and at Station S372 along the northeastern shoreline of the lake. The sublethal toxicity at these stations was seen in reduced biomass (0.26 and 0.41 mg per individual, respectively) relative to Otisco Lake (0.73 mg per individual). Chironomid emergence was affected at five locations: Stations S332, S337, S342, S344, and S354.

9.2.1.3 Comparison of 1992 and 2000 Results

Overall, the results of the 2000 42-day chronic (long term) sediment toxicity tests (based on the top 15 cm of the sediment column) and the results of the 1992 10-day acute (short term) tests (based on the top 2 cm of the sediment column) confirmed that both sub-lethal (impaired growth and reproduction) and lethal impacts (survival) are occurring in the sediments of Onondaga Lake. That is, most sediment toxicity in Onondaga Lake is confined to the nearshore zone in the southern part of the lake between Tributary 5A and Ley Creek. By contrast, little toxicity is observed elsewhere in the lake, including the deeper parts of the entire lake and the lake's eastern shore.

9.2.2 Results of Benthic Macroinvertebrate Community Evaluations

In the following sections, benthic macroinvertebrate communities in Onondaga and Otisco Lakes are compared with respect to general lakewide characteristics and station-specific characteristics.

9.2.2.1 Lakewide Comparisons of Benthic Communities in Onondaga Lake

The major characteristics of the benthic macroinvertebrate communities found in 1992 at various water depths in Onondaga and Otisco Lakes are presented in Figure 9-5. Communities were assessed in keeping with NYSDEC's evaluation (Larson, pers. comm., 1999b), because depth can substantially influence the characteristics of benthic communities (Resh and McElravy, 1993). In most cases, taxa richness and abundances of major taxa tended to decline with increasing depth in both lakes, underlining the importance of stratifying benthic communities by depth.

In Figure 9-6, the oligochaete/chironomid abundance ratios for benthic communities at various depths in Onondaga and Otisco Lakes are compared. This ratio tends to increase with increasing depth in eutrophic lakes, as more stress-tolerant oligochaetes replace chironomids at deeper depths (Wiederholm, 1980). The oligochaete/chironomid abundance ratios in Onondaga Lake exhibited the same increasing trend with increasing depth. By contrast, the oligochaete/chironomid abundance ratios in Otisco Lake showed the opposite trend (i.e., decreasing ratios with increasing depth), indicating that conditions in the deeper parts of that lake were not as stressful to benthic organisms as conditions in the deeper parts of Onondaga Lake.

9.2.2.2 Station-Specific Comparisons of Benthic Communities in Onondaga Lake

Benthic macroinvertebrate communities at stations in Onondaga Lake were compared with the communities found in Otisco Lake using several methods. Separate evaluations were conducted for the data collected in 1992 and 2000. Evaluation methods used were based on NYSDEC's methodology (NYSDEC, 1994a, 2002c) and recommendations from other benthic ecological peer review panels (PTI, 1993c; WSDE, 1996). NYSDEC (Larson, pers. comm., 1999c) and PTI (1993c), recommended that if major taxa abundance is used in the analysis additional endpoints should be measured to increase overall sensitivity. They also recommended that more than one benthic endpoint should be used to assess adverse benthic effects. Species richness and total abundance should be considered for inclusion with major taxa abundance as primary benthic endpoints.

For direct statistical comparisons univariate statistical tests (i.e., Student's t-statistic) were performed first to compare the study area and reference conditions. Subsequent analysis was more exploratory to determine the cause of the observed patterns and used multivariate techniques. Classification analysis is a multivariate technique recommended for evaluating benthic macroinvertebrate communities in the Great Lakes by the International Joint Commission (IJC, 1988). The key attributes of the approach are that it provides an integrative evaluation of all benthic taxa and has the power to detect relatively subtle patterns (IJC, 1988).

The primary method used to evaluate benthic communities was based on comparing various benthic metrics between each test station in Onondaga Lake and the appropriate reference stations in Otisco Lake. The metrics were those recommended by NYSDEC (1994a and Larson, pers. comm., 1999b) and included the following:

- **Taxa Richness:** The total number of individual taxa in a sample. The term taxa instead of species is used, as the organisms in this study are not always identified to the species level.

- **Dominance Index:** The percent composition of the three most abundant taxa.

- **Abundance of Indicator Species:** The number of non-chironomidae/ oligochaete (NCO) taxa.

- **Species Diversity:** A measure of the distribution of individuals among the taxa observed.

- **Percent Model Affinity (PMA):** A measure of similarity to a model non-impacted lake community based on percent abundance in six major groupings:

  - 20 percent Oligochaeta.
  - 15 percent Mollusca.
  - 15 percent Crustacea.
  - 20 percent non-chironomid Insecta.
  - 20 percent Chironomidae.
  - 10 percent other.

For the 1992 and 2000 benthic data, the first four metrics values for each test station were compared with the values found at the reference station using the t-test in the manner described earlier for the sediment toxicity results. For PMA, the results for each test station were compared using the model described by NYSDEC (Larson, pers. comm., 1999c).

In addition, for the 2000 benthic data, the benthic metrics were compared statistically between stations in Onondaga Lake and Otisco Lake, even though the benthic community at the shallow station in Otisco Lake was found to be dominated by a nonnative invasive species (i.e., the zebra mussel). Because few organisms were found at most of the deeper stations in Onondaga Lake, due to low levels of dissolved oxygen below the thermocline, statistical comparisons were not conducted for depths greater than 5 m.

Metrics Analysis – 1992

Five macroinvertebrate replicate samples were collected at each station, and the benthic metrics presented herein were computed by replicate. The average values of the five replicates for each metric were determined and presented for each station. The following is a discussion of how each metric was computed and how they are used to assess the health of benthic invertebrate communities.

- **Taxa Richness:** Taxa richness was determined by counting the different number of taxa per replicate (e.g., if 5 taxa are observed in a replicate, the species richness is 5). The average of the five replicates was then computed. Data from each replicate were also pooled in order to determine the total number of taxa observed at each station. The total number of taxa collected at each station was used to conduct an impairment assessment.

  If the total taxa richness at a station exceeded 32, a station was considered to be non-impaired. Total taxa richness between 25 and 32 indicated slight impairment, between 14 and 24 indicated moderate impairment, and values between 0 and 13 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Dominance Index**: The dominance index was computed by determining the total percent composition of the three most abundant species. This was performed by first determining the three taxa with the highest individual abundance in a replicate. The percent composition of these three taxa was determined by dividing the abundance (the total number of individuals of these taxa) by the total number of all individuals in the replicate.

  If the dominance index at a station was less than 61, a station was considered to be non-impaired. Dominance indices between 61 and 75 indicated slight impairment, between 76 and 90 indicated moderate impairment, and values between 91 and 100 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Abundance of Indicator Species**: The abundance of indicator species was determined by enumerating the number of taxa within each replicate that are neither Oligochaeta or Chironomidae (NCO). In general, species of oligochaetes and chironomids are tolerant of pollution stress and therefore, are not species indicative of a healthy ecosystem. As described above, data from each replicate were also pooled in order to determine the total number of indicator species observed at each station. The total number of indicator species at each station was used to conduct an impairment assessment.

If the abundance of indicator species or NCO at a station was greater than 15, a station was considered to be non-impaired. Total NCOs between 10 and 15 indicated slight impairment, between 5 and 9 indicated moderate impairment, and values between 0 and 4 signified that a station was considered to be severely impaired based on NYSDEC guidance (Larson, pers. comm., 1999b). Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Species Diversity**: Species diversity was determined using the Shannon-Wiener function as follows (Krebs, 1977):

$$H = -[(p_1)(\log_2 p_1) + (p_2)(\log_2 p_2) + \ldots]$$

where:

H = index of species diversity; and
p_i = the proportion of the sample belonging to the i^{th} taxa.

If the Shannon-Wiener diversity index at a station was greater than 3.1, a station was considered to be non-impaired. Diversity indices between 2.1 and 3 indicated slight impairment, between 1.5 and 2 indicated moderate impairment, and values less than 1.5 signified that a station was considered to be severely impaired.

- **Percent Model Affinity**: Community composition was determined using the general method described by Bode et al. (1991), including the use of a model that has been determined by the NYSDEC (Larson, pers. comm., 1999c) to be suitable for a non-impacted lake, as described above.

The percent contribution for each of the six major groups in a replicate was determined (adding up to 100 percent). For each group in the replicate, the *absolute* difference in percentage from the model value for that group was determined. The differences for each group were added per replicate. Per Bode et al. (1991), the total of the difference was multiplied by 0.5, and this number was subtracted from 100 to determine the PMA.

If the community composition or PMA at a station was greater than 64, a station was considered to be non-impaired. Model affinity between 50 and 64 indicated slight impairment, between 35 and 49 indicated moderate impairment, and values less than 35 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

Overall Assessment

The values for each of the metrics for all stations are presented in Table 9-4, including 48 Onondaga Lake (S) stations, eight tributary (T) stations, and three Otisco Lake (OT) stations. Stations were divided into the following four categories based on the results of the five metrics:

- **Non-impaired** – The macroinvertebrate community is diverse. This level of water quality includes both pristine habitats and those receiving discharges that minimally alter the biota.

- **Slightly impaired** – The macroinvertebrate community is slightly, but not significantly altered from the pristine state.

- **Moderately impaired** – The macroinvertebrate community is altered to a large degree from the pristine state.

- **Severely impaired** – The macroinvertebrate community is limited to a few tolerant species, usually midges or worms. Often only one or two species are very abundant.

Table 9-5 presents the results of the impairment assessment for each station. The cumulative review of the five metrics (referred to as a multimetric approach) was used to coalesce the metrics into a single overall assessment of each station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment category (of either non-impaired, slightly impaired, moderately impaired, or severely impaired). When less than three metrics exhibited the same impairment category, the results of all five metrics and professional judgment were used to characterize the station.

Following is a breakdown of Onondaga and Otisco Lake stations by impairment category. Tributary results are discussed in Section 9.2.2.3.

- **Non-impaired (n = 0)**: None

- **Slightly impaired (n = 12)**: Stations S26, S48, S53, S54, S67, S73, S76, S87, S100, S105, S110, and OT3.

- **Moderately impaired (n = 31)**: Stations S2, S11, S13, S14, S17, S18, S21, S34, S35, S37, S45, S46, S47, S61, S62, S72, S74, S75, S77, S82, S83, S92, S93, S94, S95, S104, S109, S111, S112, OT1, and OT2.

- **Severely impaired (n = 8)**: Stations S5, S7, S8, S22, S28, S29, S38, and S68.

Of the 51 (48 Onondaga Lake and 3 Otisco Lake) stations considered for further evaluation, none were found to be non-impaired, 12 stations were found to be slightly impaired, 31 stations were found to be moderately impaired, and 8 stations were found to be severely impaired.

Severely impaired stations are primarily located at the southern end of the lake (i.e., between Metro and Tributary 5A). One station (Station S68) considered to be severely impaired is located near Wastebeds 1 to 8. Moderately impaired stations are found throughout the lake.

Statistical and Classification Analysis (1992)

The results of the statistical analysis of four of the five benthic metrics requested by NYSDEC are summarized in Table 9-6 and Figures 9-7, 9-8, 9-9, 9-11, 9-12, 9-13, and 9-14. The results of the metrics analysis show that NCO richness and diversity were the most sensitive metrics, since they identified 48 and 28 stations as being impacted, respectively. By contrast, dominance was much less sensitive, identifying only 8 stations as being impacted. Taxa richness was intermediate in sensitivity, identifying 25 stations as impacted.

The patterns described above indicate that much of the littoral zone less than 4.5 m deep in Onondaga Lake is impacted. These patterns of impacted stations are correlated when one examines them in conjunction with the 1992 and 2000 sediment toxicity test results. As stated above, these tests indicate sub-lethal and lethal effects in nearshore sediments. The most useful and discriminating benthic metric appears to be taxa richness, which showed no depth-related bias and produced patterns similar to those based on sediment chemistry and sediment toxicity.

Using classification analysis and further segmenting the 4.5 m interval into 1.5 m and 4.5 m for easier review of the results, three benthic groups were identified in the 1.5 m depth stratum (Figure 9-13) and two benthic groups were identified in the 4.5-m depth stratum sampled in 1992 (Figure 9-14). For the 1.5 m depth stratum, based on similarity to stations from Otisco Lake, benthic communities from stations in Group A could be considered slightly altered; communities from Group B were considered moderately altered; and communities from Group C were considered to exhibit major alterations. For the 4.5 m depth, based on similarity to stations from Otisco Lake, benthic communities from stations in Group A were considered slightly altered, and communities from Group B were considered to exhibit major alterations.

The various benthic groups identified by the classification analysis are compared with respect to major community variables in Figure 9-15. In most cases, mean taxa richness and mean abundances of major benthic taxa for the minimally altered group were considerably greater than the respective mean values for the group exhibiting major alterations. In addition, values of taxa richness and amphipod abundance for the moderately altered group at 1.5 m depth were intermediate in magnitude. These patterns indicate that the major characteristics of benthic communities corresponded to the results of the classification analysis.

The spatial distribution of the various kinds of benthic effects described above is presented in Figure 9-16. Major alterations of benthic communities were found in two nearshore areas: off Tributary 5A and in the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume). Moderate alterations of benthic communities were found at most of the remaining nearshore stations in the southern part of the lake between Tributary 5A and Ley Creek.

Metrics Analysis – 2000

The five benthic metrics are summarized in Table 9-7 for the August 2000 data, including nine Onondaga Lake (S) stations and one Otisco Lake (OT) station. Onondaga Lake and Otisco Lake stations that were located in water depths greater than 5 m were excluded from the assessment (although NYSDEC's May 27, 1999 letter indicates stations deeper than 3 m were potentially depth-impacted, this report includes the stations at the 5 m depth to provide greater spatial coverage of the lake; thus, six of the 15 Onondaga Lake stations and one of the two Otisco Lake stations sampled in 2000 were excluded [Larson, 1999b, pers. comm.]). Analysis of the 1992 data on benthic communities excluded all stations below 4.5 m. However, no reference station (i.e., Stations OT-6 or OT-7) was shallower than 4.5 m in the 2000 sampling event. The Otisco Lake stations were at depths of 5 and 9 m, respectively. As such, the limit of exclusion was increased to 5 m in order to permit the use of a reference station for analysis.

The information presented below is a brief summary of the data analyses, as per the five matrices specified by NYSDEC. The reference value(s) are from Otisco Lake's Station OT-6.

- **Taxa Richness:** Values for all but one station (Station S372) in Onondaga Lake were lower than the reference value of 19, with values ranging from 8 to 16.

- **Dominance:** Values at seven of the nine Onondaga Lake stations were greater than the reference value of 77 percent. Dominance values ranged from 63 to 91 percent.

- **NCO Taxa:** Values for all Onondaga Lake stations were considerably lower than the reference value of 9, with values ranging from <1 to 4.

- **Community Composition (PMA):** Values at all of the Onondaga Lake stations were less than the reference value of 60, ranging from 22 to 54.

- **Species Diversity:** Values at seven of the nine Onondaga Lake stations were lower than the reference value of 2.4. Diversity at Onondaga Lake Stations S365 (2.5) and S372 (3.1) were higher than the reference value.

Table 9-8 presents the results of the impairment assessment for each station. As for the 1992 data, the cumulative review of the five metrics (referred to as a multimetric approach) was used to coalesce the metrics into a single overall assessment of each station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment category (of either non-impaired, slightly impaired, moderately impaired, or severely impaired). When less than three metrics exhibited the same impairment category, the results of all five metrics and professional judgement were used to characterize the station. Following is a breakdown of stations by impairment category:

- **Non-impaired (n = 0):** None.

- **Slightly impaired (n = 3):** Stations S365, S372, and OT-6.

- **Moderately impaired (n = 6):** Stations S305, S323, S332, S337, S342, and S344.

- **Severely impaired (n = 1):** Stations S317.

Of the 10 (9 Onondaga Lake and 1 Otisco Lake) stations considered for further evaluation, none were found to be non-impaired; 3 stations were found to be slightly impaired; 6 stations were found to be moderately impaired; and 1 station was found to be severely impaired.

The severely impaired station (Station S317) is located in the southern end of the lake between the Metro outfall and the mouth of Onondaga Creek. Moderately impaired stations are found throughout the lake, clustered between Tributary 5A and Harbor Brook, and near the mouths of Ninemile Creek and Ley Creek.

Two of the three slightly impaired stations are located in Onondaga Lake: one (Station S365) is north of the mouth of Tributary 5A, and the other (Station S372) is in the northwestern portion of the lake. The one reference station found to be slightly impaired (Station OT-6) is located in Otisco Lake. This reference station (OT-6) differed considerably from seven of the nine Onondaga Lake stations, as it possessed a disproportionately high number of zebra mussels (*Dreissena polymorpha*). The zebra mussel comprised 28 percent of the total individuals for Station OT-6. In Onondaga Lake, the zebra mussel was rarely observed in 1992, but was found in abundance at two stations in 2000. At Stations S365 and S372, the zebra mussel comprised 25 and 7.5 percent of the benthic populations, respectively (based on the average of five replicates).

However, despite the presence of zebra mussels, an important difference between the benthic assemblages at Otisco and Onondaga Lakes are the numbers and percentages of NCO taxa to chironomidae and oligochaete taxa. At Station OT-6, approximately half of the total number of taxa (15 of 33 taxa), and half the taxa richness (9 of 19 taxa), were comprised of NCO taxa. Comparatively, in Onondaga Lake, the NCO to chironomidae and oligochaete ratio was much lower for eight of the nine stations. In fact, at seven of the eight Onondaga Lake stations, the taxa were comprised of 25 percent or less NCO taxa. This is important to acknowledge because, as stated earlier, chironomidae and oligochaete taxa are generally considered more pollution-tolerant than NCO taxa.

Stations S365 and S372 had the two highest diversity readings, at 2.5 and 3.1, respectively, and the two highest total number of taxa for Onondaga Lake. However, these stations also had high NCO taxa to chironomidae and oligochaete ratios for total taxa and taxa richness. The ratios of NCO taxa to chironomidae and oligochaete taxa for these two stations are the following:

- **Total Taxa:** Stations S365 (4 NCO of 26 total taxa) and S372 (6 NCO of 36 total taxa).

- **Taxa Richness:** Stations S365 (3 NCO of 14 total taxa) and S372 (4 NCO of 22 total taxa).

Thus, the high diversity of these two stations may not be indicative of a healthy environment, as a large majority of these taxa are pollution-tolerant.

Station S342 was the only Onondaga Lake station to exhibit somewhat similar NCO to chironomidae and oligochaete ratios as the reference station OT-6.

In the less than 5 m depth stratum in Onondaga Lake in 2000, the following general patterns were found:

- **Taxa Richness:** Values for all but one station in Onondaga Lake were lower than the reference value of 33 total taxa. Taxa richness at Station S372 (36 total taxa) off the northeastern shoreline of the lake exceeded the reference value.

- **NCO Taxa:** Values for all Onondaga Lake stations were considerably lower than the reference value of 15 total taxa, with 8 of the 9 stations having a value of less than half the reference value.

- **Species Diversity:** Values at most stations in Onondaga Lake were lower than the value of 2.4 for Otisco Lake. Diversity at Stations S365 (2.5) and S372 (3.7) were higher than the reference value.

- **Dominance Index:** Values at most stations in Onondaga Lake were greater than the reference value of 77 percent, most likely reflecting the dominance of oligochaetes and chironomids at most stations in Onondaga Lake.

- **Percent Model Affinity:** Values at most stations in Onondaga Lake were less than 30 percent with the zebra mussel included in the analysis with Otisco Lake having the highest PMA of 60 percent.

Statistical and Classification Analysis (2000)

The statistical analysis results for four of the five benthic metrics requested by NYSDEC are summarized in Table 9-9. Results of the metrics analysis show that NCO richness and species richness (total taxa) were the most sensitive metrics, since they identified nine and eight stations as being impacted, respectively. By contrast, dominance was much less sensitive, identifying no station as being impacted. Species diversity was intermediate in sensitivity, identifying five stations as impacted.

The patterns described above indicate that much of the littoral zone less than 4.5 m deep in Onondaga Lake is impacted. These effects are corroborated when one examines them in conjunction with the 1992 and 2000 sediment toxicity test results and 1992 benthic metrics. As stated above, these tests indicate sub-lethal and lethal effects in nearshore sediments.

For the nine shallow stations (i.e., 1.5 to 5 m) sampled in Onondaga Lake in 2000, one group of closely related stations was identified (Figure 9-17), consisting of the six shallow stations that extended from south of Tributary 5A to Ley Creek (i.e., Stations S317, S323, S332, S337, S342, and S344). The remaining shallow stations in Onondaga Lake did not cluster closely with other lake stations, most likely because they were from different parts of the lake: north of Tributary 5A (Station S365), off Ninemile Creek (Station S305), and off the northeastern shoreline (Station S372).

The reference station from Otisco Lake (Station OT6) showed little similarity to the stations from Onondaga Lake. Inspection of the taxonomic composition of the reference station showed that the benthic community at that station was dominated by the zebra mussel, which was not found in Otisco Lake during the RI sampling in 1992. That species accounted for nearly half (i.e., 45 percent) of the total number of organisms found at Station OT6 in 2000, at a density of 23,000 individuals/m^2.

In Onondaga Lake, the zebra mussel was rarely observed in 1992, but was found in abundance at two stations in 2000. At Station S365 (north of Tributary 5A) the mussel comprised 31 percent of the benthic community, at a density of 7,600 individuals/m^2. At Station S372, off the northeastern shoreline, the mussel comprised 12 percent of the benthos, at a density of 5,400 individuals/m^2. The large abundances of zebra mussels at those two stations was likely one reason that the stations were not similar to other stations in the lake, based on the results of the classification analysis.

9.2.2.3 Comparisons of Benthic Communities in Tributaries of Onondaga Lake

In 1992, the mouths of the eight tributaries were sampled for evaluation of their benthic community structure (see Chapter 7, Figure 7-2). Table 9-10 presents the results of the impairment assessment for each tributary station. The cumulative review of the five metrics was used to coalesce the metrics into a single overall assessment of each station, as was done for the lake. Following is a breakdown of stations by impairment category:

- **Non-impaired (n = 2):** Stations T11 (Bloody Brook) and T15 (Sawmill Creek).
- **Slightly impaired (n = 0):** None.
- **Moderately impaired (n = 2):** Stations T13 (Ninemile Creek) and T7 (East Flume).
- **Severely impaired (n = 4):** Stations T1 (Harbor Brook), T3 (Onondaga Creek), T5 (Ley Creek), and T9 (Tributary 5A).

The benthic macroinvertebrate communities near the mouths of the tributaries to Onondaga Lake were compared using the same methods of classification analysis described above for benthic communities in the lake. The analysis identified three groups of tributaries based on abundances of benthic macroinvertebrates (Figure 9-18), as follows:

- Group A included the four largest tributaries (Harbor Brook, Onondaga Creek, Ley Creek, and Ninemile Creek).
- Group B included the two small tributaries on the western shoreline of the lake (the East Flume and Tributary 5A).
- Group C included the two small tributaries on the eastern shoreline of the lake (Bloody Brook and Sawmill Creek).

The major characteristics of the benthic macroinvertebrate communities near the mouths of the eight tributaries are presented in Figure 9-1. The following major differences were found among the three groups of tributaries identified in the classification analysis:

- **Taxa richness** – This variable was highest in Group C, intermediate in magnitude in Group A, and generally lowest in Group B.
- **Oligochaetes** – The highest oligochaete densities were found in Group A and Tributary 5A, whereas densities in most of the other tributaries were uniformly lower.

- **Chironomids** – The highest chironomid densities were found in Group C, the East Flume, and Ninemile Creek, whereas densities in the other tributaries were highly variable, but generally much lower.

- **Amphipods** – The highest amphipod densities were found in Group C and Ninemile Creek, whereas densities in the other tributaries were uniformly lower and no amphipods were found in the East Flume and Tributary 5A.

These results indicate that the major characteristics of benthic communities were generally similar within the three groups of tributaries identified by classification analysis and generally different among the three groups. Although tributary size and shoreline location may have been partly responsible for the patterns identified by the classification analysis, the tributary groupings may also have been influenced by stressors such as chemical toxicity and organic enrichment.

The high taxa richness in Group C suggests that communities in those tributaries (i.e., Bloody Brook and Sawmill Creek) are minimally altered. High taxa richness indicates that many less-abundant species inhabit those tributaries. In many cases, the less-abundant benthic species tend to be more sensitive to stressors than the more-abundant species.

In contrast with the patterns described above for benthic communities in Group C, the relatively low taxa richness in Group B suggests that communities in those tributaries (i.e., the East Flume and Tributary 5A) are altered to a much greater degree. However, the moderate densities of chironomids found in the East Flume suggest that communities in that tributary are less altered than communities in Tributary 5A.

The intermediate values of taxa richness found for Group A suggest that communities in those tributaries (i.e., Harbor Brook, Onondaga Creek, Ley Creek, and Ninemile Creek) are moderately altered. In addition, the high densities of oligochaetes in Group A suggest that communities in those tributaries may be substantially affected by chemical contamination, ionic waste, or organic enrichment. However, the relatively high densities of chironomids and amphipods, as well as the relatively low densities of oligochaetes in Ninemile Creek, indicate that communities in that tributary are much less altered than communities in the other tributaries from Group A.

However, if the results of the impairment assessment are examined, a slightly different understanding emerges when all benthic metrics are considered, as required by NYSDEC and suggested by several peer review panels. The Group C tributaries, Sawmill Creek and Bloody Brook, are non-impaired, while the majority of Group A tributaries (with the exception of Ninemile Creek but including Tributary 5A) are severely impaired, and Group B (without Tributary 5A but with Ninemile Creek) are moderately impaired. However, when the densities of chironomids, amphipods, and oligochaetes are examined (Figure 9-19), it can be seen that Tributary 5A resembles Harbor Brook, Ley Creek, and Onondaga Creek, while Ninemile Creek may be more altered than communities in the other Group A tributaries and more in line with the East Flume benthic communities.

However, NYSDEC (Larson, 1999b, pers. comm.) stated that, based on NYSDEC kick sampling of the tributaries to Onondaga Lake in 1989 and from 1994 to 1996, the data indicate that Bloody Brook and Sawmill Creek are at least moderately impaired. This raises concerns about using a lake model for the tributary mouths. More appropriate assessments are obtained based on the kick-sampling results (Larson, 1999b, pers. comm.), which indicate that Harbor Brook, Ley Creek, Bloody Brook, Ninemile Creek, and Sawmill Creek are moderately impacted and that Onondaga Creek, the East Flume, and Tributary 5A are severely impacted.

9.2.3 Comparison of Results of Sediment Toxicity Test and Benthic Macroinvertebrate Community Evaluations

In this section, the 1992 results of the sediment toxicity tests and benthic macroinvertebrate community evaluations for Onondaga Lake are compared to determine the extent to which they agree. Close agreement between these different kinds of indicators enhances confidence that the observed patterns of adverse biological effects are real and that they are likely the result of chemical toxicity.

9.2.3.1 Comparisons Based on Benthic Groups

The sediment toxicity results for stations in the various benthic groups identified by classification analysis are compared in Figures 9-20 and 9-21. In general, the toxicity results(mean survival and mean biomass) were closely related to the benthic groups in Figures 9-13 and 9-14, with mean survival and mean biomass for both amphipods and chironomids generally declining from groups based on minimally altered benthic communities (based on benthic metrics) to groups based on communities exhibiting major alterations (based on benthic metrics).

9.2.3.2 Comparisons Based on Adverse Effects

Comparisons of results of the 1992 effects designations (i.e., the presence or absence of adverse biological effects) based on the sediment toxicity tests and the benthic macroinvertebrate community evaluations are presented in Figure 9-22. The percentages of stations identified as having adverse effects varied for the five biological indicators as described below:

- The lowest values were found for amphipod survival and chironomid biomass (0.8 and 13 percent, respectively).
- The highest values were found for benthic community alterations and chironomid survival (43 and 29 percent, respectively).

These results indicate that chironomid survival and benthic community alterations were the most sensitive indicators of sediment toxicity. Chironomid survival would be expected to be a sensitive indicator because it is a response of an organism that burrows into the sediment. Benthic community alterations would also be expected to be a sensitive indicator because it incorporates chronic exposure and sublethal effects on resident organisms, as well as acute exposure and lethal effects on those organisms.

Most stations at which lethal sediment toxicity and major benthic community alterations were found are located in the nearshore zone between Tributary 5A and the Metro outfall (Figure 9-22). Although sublethal toxicity was found throughout most of the southern half of the lake (including areas far from shore), most of the widespread sublethal toxicity was based only on the chironomid test and may be due to factors such as substrate type.

Agreement between the effects designations based on sediment toxicity tests and benthic community alterations was very high. Of the 48 stations at which both kinds of indicators were evaluated, the two kinds of indicators agreed on effects designations in 28 cases (58 percent) and disagreed in 20 cases (37 percent). This level of agreement was significant ($P \leq 0.01$, binomial test) compared to an assumed level of random agreement of 50 percent.

Based on the 2000 toxicity test results of the nine stations that were evaluated for both toxicity and benthic metrics, eight stations showed toxic and benthic community impairment with six of eight stations being moderately to significantly impaired.

9.2.4 Development of Site-Specific Sediment Effect Concentrations and Consensus Probable Effect Concentrations

Sediment effect concentrations (SECs) and consensus probable effect concentrations (PECs) were derived (using the sediment chemistry and toxicity data collected in 1992 and 2000) to allow site-specific assessment of whether the sediment chemical concentrations found at various stations in the lake were potentially related to adverse biological effects.

The SECs and PECs were developed primarily using the 1992 toxicity test data because that data set contained a large number of stations (i.e., 79) that were distributed across broad ranges of sediment chemical of concern (COC) concentrations throughout the entire lake. Development of SECs using the smaller 2000 data set was performed to evaluate whether the chronic toxicity endpoints would provide a different outcome than the short-term tests performed in 1992. Onondaga Lake SECs and PECs were developed for all thirteen metals and 17 organic contaminant compounds identified as COCs in Chapter 6. In addition, total polychlorinated biphenyls (PCBs) and total PAHs were broken out into specific Aroclor components and individual polycyclic aromatic hydrocarbon (PAH) compounds to provide additional detail for the risk characterization in Chapter 10.

The information on benthic macroinvertebrate communities collected during 1992 was not used to develop the SECs or PECs, because it was found that benthic communities at every station in the lake are impaired to some degree; thus, it is not possible to calculate an SEC because these calculations require that a certain proportion of stations have no effects. The results of the benthic macroinvertebrate evaluations for Onondaga Lake were, therefore, used primarily to interpret the magnitude and significance of any potential sediment toxicity predicted on the basis of the SECs. Site-specific SECs were developed for Onondaga Lake using the apparent effects threshold (AET) approach, as well as calculation of effects range-low (ER-L), effects range-median (ER-M), probable effect levels (PEL), and threshold effects level (TEL) values. Consensus-based PECs for COCs in Onondaga Lake were developed following the methodology described in MacDonald et al. (2000) and Ingersoll et al. (2000) as the geometric mean of the site-specific SECs. These sediment guidelines, coupled with other site-specific information on potential risks to ecological receptors, may be used as one tool in the derivation of site-specific sediment cleanup criteria in the FS. USEPA (1997b) recently used AETs (in conjunction with other kinds of SECs) to evaluate the potential toxicity of sediments from over 21,000 stations throughout the US as part of the National Sediment Quality Survey. The AET approach has also been used by the Washington State Department of Ecology (WSDE) to develop promulgated state sediment standards for managing contaminated sediment in Puget Sound, Washington (WSDE, 1995).

Based on recent reviews of the method and development of the proposed Freshwater Sediment Guidelines by WSDE (1997), Ingersoll et al. (1996, 2000) indicated that SEC values based on dry weight organic chemical concentrations either outperform or are not significantly different than organic carbon-normalized data in sensitivity (i.e., false negatives) and efficiency (i.e., false positives). In this BERA, all SECs are developed based on sediment dry weight contaminant concentrations for both organic and inorganic contaminants.

9.2.4.1 Development of Apparent Effect Threshold Effect Levels

The AET for a given chemical is the sediment concentration above which a particular adverse biological effect (e.g., increased mortality or decreased biomass) is always statistically significant ($P \leq 0.05$) relative to appropriate reference conditions (WSDE, 1997). The objective of the AET approach is to identify concentrations of contaminants that are associated exclusively with sediments exhibiting statistically significant biological effects relative to reference sediments.

A detailed description of AET methodology is found in Michelsen and Shaw (1996). AETs can be developed for any kind of biological indicator that has corresponding information on sediment chemical concentrations.

In order to conduct an appropriate analysis, sites were grouped, or "matched," according to a sediment characteristic (e.g., grain size, total organic carbon [TOC], or water depth). Sediment type was utilized to assign the appropriate reference station to the Onondaga Lake stations. Station OT3 is primarily made up of sand, and the next least-impacted site, Station OT4, is primarily made up of fines. Michelsen and Shaw (1996) offers guidance for samples collected from multiple reference stations. Pair-wise comparisons are necessary because each site station needs to be handled separately; therefore, multiple comparison tests that compare the distribution of the data for all locations are not appropriate (Michelsen and Shaw, 1996).

For each chemical, AETs were developed for all four measures of sediment toxicity from the 1992 toxicity data (amphipod survival and biomass and chironomid survival and biomass) and six measures of toxicity from the 2000 data set evaluated during the RI (i.e., amphipod survival, biomass, and reproduction and chironomid survival, biomass, and emergence) in Tables 9-11 and 9-12. The 2000 AETs in Table 9-12 are provided for comparison purposes only and are not used to derive site-specific SECs.

The final AET for each COC was defined as the lowest of all four AETs (amphipod survival and growth and chironomid survival and growth) derived from the 1992 toxicity data set. AETs could not be determined for four organic compounds (i.e., pyrene, indeno(1,2,3-cd)pyrene, chrysene, and benz[a]anthracene) because the concentrations of those COCs were not found over a sufficiently large range.

9.2.4.2 Development of Other Site-Specific Sediment Effect Concentrations

Two commonly used approaches to developing SECs, other than the AET approach, were evaluated for site-specific application to Onondaga Lake. These approaches are currently used by the National Oceanic and Atmospheric Administration's (NOAA's) National Status and Trends Program to evaluate sediments nationwide; by Environment Canada (CCME, 1995); and by the State of Florida (MacDonald, 1994) to derive sediment quality guidelines.

One approach was developed by Long and Morgan (1990) and calculates two kinds of SECs for each chemical, the ER-L and ER-M. The second approach was developed by MacDonald et al. (1996) and also calculates two kinds of SECs for each chemical, the TEL and PEL. For both approaches, the two SECs represent a lower level (i.e., ER-L and TEL) below which adverse effects are not expected, and a higher level (i.e., ER-M and PEL) above which effects are likely to occur. The approaches of Long and Morgan (1990) and MacDonald et al. (1996) calculate SECs as follows:

- **ER-L:** 10^{th} percentile of the concentration distribution for the effects data.
- **ER-M:** Median of the concentration distribution for the effects data.
- **TEL:** Geometric mean of the 15^{th} percentile of the concentration distribution for the effects data and the median of the distribution for the no-effects data.
- **PEL:** Geometric mean of the ER-M and the 85^{th} percentile of the concentration distribution for the no-effects data.

For both approaches, the effects distribution for each chemical is defined as those stations at which a biological effect is observed and the associated chemical concentration is greater than or equal to twice the mean concentration of the no-effect stations. In addition, MacDonald et al. (1996) stipulate that it is desirable for both the effects and no-effects distributions to include at least 20 data entries.

A major distinction between the various kinds of SECs is the manner in which effects and no-effects data are used. As shown by the definitions above, the ER-L/ER-M values are based only on effects data, whereas the TEL/PEL values are based on both the effects and no-effects data. As described previously in this section, AET values are based only on the no-effects data (i.e., "nonimpacted" stations).

For Onondaga Lake, the various kinds of SECs were developed primarily on the basis of the chironomid survival endpoint, which identified effects at 35 of the 79 stations sampled in 1992. None of the other endpoints identified effects at a sufficient number of stations to achieve the stipulation of MacDonald et al. (1996) that the effects and no-effects distributions should both have at least 20 data entries. The results of these calculations are presented for comparison in Table 9-13.

The various SECs were initially calculated on the basis of chironomid survival only and on the basis of any kind of toxic effect. Because the resulting SECs showed little differences, the subsequent analyses were conducted only using the survival endpoint.

9.2.4.3 Evaluation of Mercury Sediment Effect Concentrations

The results of the mercury SEC comparisons are presented in Table 9-13. To assess the accuracy with which the various sets of SECs identified the presence or absence of effects in Onondaga Lake in 1992, the following performance criteria were calculated using mercury as an example:

- **False Positives (Type I Error):** The percentage of stations predicted to have effects (i.e., based on exceedance of one or more of the SECs) that actually had no observed effects based on the chironomid survival results.

- **False Negatives (Type II Error):** The percentage of stations predicted to have no effects (i.e., based on lack of exceedance of any of the SECs) that actually had observed effects based the chironomid survival results.

- **Overall Accuracy:** The percentage of all samples that were correctly predicted to have effects, or not to have effects based on the SECs.

From a practical standpoint, a high percentage of false positives is undesirable for a set of SECs because a large number of stations predicted to have toxic sediments actually would not have such sediments. This could potentially result in remediation of areas where such activities are not warranted. By contrast, a high percentage of false negatives is undesirable because a large number of stations predicted not to have toxic sediments actually would have such sediments. This could potentially result in remedial actions not being selected for all areas where they are warranted. Ideally, therefore, a set of SECs should have relatively small percentages of both false positives and false negatives.

The major performance criteria patterns for mercury are as follows:

- **False Positives (Type I Error):** The AET for mercury had the lowest false positive error (14 percent), whereas values for the other SECs ranged from 33 percent (ER-M and PEL), 53 percent(ER-L), and 48 percent (TEL).

- **False Negatives (Type II Error):** The ER-L had the lowest false positive error (12 percent), whereas errors for the other SECs were 47 percent (ER-M and PEL), 48 percent (TEL), and 82 percent (AET).

- **Overall Accuracy:** The mercury ER-M/PEL had the highest degree of overall accuracy (66 percent), followed by the AET value of 65 percent, with the TEL at 56 percent and the ER-L at 51 percent.

Based on the results of the SEC evaluations described above, it can be concluded that no one of the methodologies employed accurately describe or predict threshold concentrations of toxicity in Onondaga Lake sediments, nor can any one methodology accurately attribute the toxicity observed to any single contaminant. These values cannot be absolute because of the exposure of organisms to a complex mixture of metals and other contaminants which make it difficult to attribute the toxicity to any particular contaminant. However, collective evaluation through a strength-of-evidence approach does provide useful information.

9.2.4.4 Evaluation of Sediment Effect Concentrations Based on the 2000 Data

As described previously, sediment toxicity data were collected at 15 stations in Onondaga Lake in 2000 primarily to compare the results of the 42-day amphipod and chironomid toxicity tests with the 1992 results of the 10-day amphipod and chironomid toxicity tests. As shown in Section 9.2.1.3, the results of the 42-day tests were similar to those of the 10-day tests with respect to the areas of the lake in which sediment toxicity was present or absent.

In this section, an evaluation is conducted to determine whether SECs based on the 2000 data are similar to those developed using the 1992 data. However, because the 2000 data were collected at a relatively small number of stations, they will be considered for use in risk characterization in a qualitative manner. In addition, because the concentration ranges of the organic chemicals measured at the 15 stations were not evenly distributed between high, medium, and low values (as opposed to mercury), it was only possible to develop meaningful AETs for metals. Organic AETs were calculated (Table 9-12), but are provided only for qualitative comparison to the 1992 AET values.

The results of the 1992/2000 SEC comparisons are presented in Table 9-14 for seven metals. There was no consistent pattern with respect to one kind of SEC being greater than the other. The two kinds of AETs generally agreed, within a factor of two, for all of the metals; therefore, agreement between the two sets of AETs can be considered relatively good. For example, Long et al. (1995) considered agreement among various kinds of SECs to be close when they agreed within a factor of three.

9.2.4.5 Development of Consensus Based Probable Effect Concentrations

Consensus-based probable effect concentrations (PECs) for COCs in Onondaga Lake were developed to support an assessment to sediment-dwelling organisms and follow the methodology described in MacDonald et al. (2000) and Ingersoll et al. (2000). The PECs are the geometric mean of the AET, PEL, TEL, ER-M, and ER-L SECs. In addition, the PECs:

- Provide a unifying synthesis of site-specific effects concentrations.
- Reflect causal rather than correlative effects.
- Account for the effects of sediment COCs.

The PECs do not consider the potential for:

- Bioaccumulation in aquatic species.
- Potential effects that could occur throughout the food web as a result of bioaccumulation.
- Synergistic or antagonistic effects of chemical mixes in the sediment.

Onondaga Lake PECs were developed for all compounds identified as COCs (see Chapter 6) based on the 1992 data and are presented in Table 9-13.

9.2.5 Acid-Volatile Sulfide

Based on the concentrations of acid-volatile sulfide (AVS) observed throughout Onondaga Lake during the 1992 and 2000 RI sampling, the bioavailability of divalent metals such as cadmium, copper, lead, mercury, nickel, silver and zinc should be limited during the summer months in anoxic sediments. Because AVS binds with metals, it reduces their bioavailability (DiToro et al., 1990, 1992). When the molar ratio of simultaneously extracted metals (SEM) to AVS is less than or equal to 1, toxicity due to the divalent metals is not predicted because a sufficient amount of AVS is present to bind with the total amount of SEM. However, when the SEM/AVS ratio is greater than 1, toxicity may occur, depending on the concentrations of SEM and the presence or absence of other factors that modify the bioavailability of metals (USEPA, 1994a, 1995a; Ankley et al., 1996; Berry et al., 1996). Uncertainties related to use of the SEM/AVS ratio are discussed in Chapter 10.

As shown in Figure 9-23, AVS concentrations throughout most of the deeper parts of Onondaga Lake in 1992 were very high (>2,000 mg/kg), reflecting the hypereutrophic condition of the lake. By contrast, concentrations in most of the shallow nearshore areas of the lake were less than 500 mg/kg. The

SEM/AVS ratios at most stations in the lake were less than or equal to 1 during both 1992 and 2000 (Figure 9-24), largely reflecting the high concentrations of AVS found throughout most of the lake. The SEM/AVS ratios were greater than 1 at 13 stations in the shallowest parts of the lake in 1992, indicating that divalent metals could cause sediment toxicity at those stations. However, most of those stations were characterized by coarse-grained sediments with low concentrations of both SEM and AVS. Therefore, SEM/AVS ratios greater than 1 at those stations would not necessarily result in sediment toxicity. In fact, sediment toxicity was observed at only 3 of those 13 stations during the RI.

The theoretical wisdom indicates that there should not be any methylmercury in the sediments of the hypolimnion if there is excess AVS. As indicated on Figure 9-24, there is (or should be) enough AVS to complex the divalent metals and that should bind the inorganic mercury so it is not available to form methylmercury. There is sufficient literature to support the contention that AVS will bind inorganic mercury and make it unavailable for methylation. On the other hand, Long et al. (1998) suggests this process does not always occur under field conditions. Methylmercury formation in aquatic systems is influenced by a wide variety of environmental factors. The efficiency of microbial mercury methylation generally depends on factors such as microbial activity and the concentrations of bioavailable mercury, which in turn are influenced by temperature, pH, redox potential, and the presence of inorganic and organic complexing agents. Earlier studies noted that mercury methylation is inhibited by high sulfide levels in soils, sediments, and bacterial cultures (Fagerstrom and Jernelov, 1971; Bisogni and Lawrence, 1975; Jacobs and Keeney, 1974; Talmi and Mesmer, 1975). It was speculated that in the presence of sulfide, Hg forms insoluble HgS, which is not readily available for methylation under anaerobic conditions (Fagerstrom and Jernelov, 1971; Gillespie, 1972). However, current studies have reported that the solubility of Hg is actually increased in the presence of excess sulfide, most likely due to the formation of soluble complexes (Gognon et al, 1997; Benoit et al, 1998; Bloom et al., 1999). Recently, the work of Benoit et al (1998, 1999a, 1999b) shows that sulfide affects the bioavailability of mercury by controlling mercury speciation, and suggests that the bioavailability of mercury in sediments is determined by the concentration of neutral dissolved mercury complexes such as HgS^0, which may readily diffuse across bacterial cell membranes.

It is also important to note that the use of AVS to predict non-toxic sediment is less conservative than the use of sediment quality guidelines. Long et al. (1998) found that AVS resulted in a 19 percent false negative rate when they evaluated 77 samples from five marine locations. The authors reported that the use of the NOAA ER-Ls resulted in no false negatives. AVS may also be a poor predictor of bioaccumulation of metals (Ankley, 1996). Work by Howard and Evans (1993) in stratified lakes in Canada points toward another issue: seasonal changes of AVS concentrations in eutrophic lakes where significant temporal and spatial changes in AVS concentrations occur and bioavailability of divalent metals may increase strongly during times when the lake sediments are oxidized. Since no temporal variation in AVS levels was determined in this BERA, the usefulness of these data to predict the lack of year-round bioavailability and toxicity is questionable.

9.3 Effects Characterization for Terrestrial and Aquatic Vertebrates

9.3.1 Selection of Measures of Effects

For the selection of TRVs in this assessment, a comprehensive literature search of laboratory and field studies was conducted on the toxicity of COCs to terrestrial and aquatic vertebrates. Using the Ovid search engine, a variety of databases were searched for references containing toxicity information, including the following:

- TOXLINE.
- TOXNET (including the Aquatic Information Retrieval Database [AQUIRE]).
- USEPA's and US Army Corps of Engineers' (USACE's) Environmental Residue Effects Database (ERED).
- National Library of Medicine (NLM) MEDLINE.

Secondary sources that were used to identify studies that may have been overlooked in the database searches included the following:

- US Fish and Wildlife Service Contaminant Hazard Reviews.
- Agency for Toxic Substances Disease Registry (ATSDR) documents.
- USEPA Great Lakes Water Quality Initiative documents.
- Jarvinen and Ankley database (1999).

A number of criteria were considered in order to evaluate the appropriateness of a particular study for inclusion in the database used for this BERA. First of all, doses should be quantified and reported. An appropriate study design, including the use of adequate sample size and an appropriate negative control group, should be included in the design. Appropriate statistical analyses should be conducted and the statistical significance of the results reported. The remainder of this section describes the rationale that was used to select TRVs for the representative receptors.

Some studies examine toxicity endpoints (such as lethality, growth, and reproduction) that are thought to have greater potential for adverse effects on populations of organisms than toxicity endpoints evaluated in other studies. Other studies examine toxicity endpoints, such as behavior, disease, cell structure, or biochemical changes, that affect individual organisms but may not result in adverse effects at the population level. For example, toxic effects such as enzyme induction may or may not result in adverse effects to individual animals or populations. This BERA prefers TRVs from studies that examine the effects of COCs on growth or reproduction, as these endpoints typically present the greatest risk to the viability of the individual organism and, therefore, survival of the population. Thus, these are considered to be the endpoints of greatest concern relative to the stated assessment endpoints.

Because of the persistence of contaminants in Onondaga Lake, the exposure of ecological receptors is expected to be long-term. Some reproductive effects of contaminants are typically seen after long-term exposure, or in offspring of exposed individuals. Therefore, studies of chronic exposure were used to select TRVs for this risk assessment.

Dose-response studies compare the response of organisms exposed to a range of doses to that of a control group. Ideally, doses that are below and above the threshold level that causes adverse effects are examined. Toxicity endpoints determined in dose-response and other studies include:

- **No observed adverse effect level (NOAEL):** The highest exposure level shown to be without adverse effect in organisms exposed to a range of doses. NOAELs may be expressed as dietary doses (e.g., mg COC consumed/kg body weight per day [-d]), as concentrations in external media (e.g., mg COC/kg food), or as concentrations in tissue of the exposed organisms (e.g., mg chemical/kg egg).

- **Lowest observed adverse effect level (LOAEL):** The lowest exposure level shown to produce adverse effect in organisms exposed to a range of doses. LOAELs may also be expressed as dietary doses (e.g., mg COC consumed/kg body weight-d), as concentrations in external media (e.g., mg COC/kg food), or as concentrations in tissue of the exposed organisms (e.g., mg chemical/kg egg).

- **LD_{50}:** The lethal dose that results in the death of 50 percent of the exposed organisms. Expressed in units of dose (e.g., mg COC administered/kg body weight of test organism-d).

- **LC_{50}:** The lethal concentration in some external media (e.g. food, water, or sediment) that results in the death of 50 percent of the exposed organisms. Expressed in units of concentration (e.g., mg COC/kg wet weight [ww] food).

- **ED_{50}:** The effective dose that results in a sublethal effect in 50 percent of the exposed organisms (mg/kg-d).

- **EC_{50}:** The effective concentration in some external media that results in a sublethal effect in 50 percent of the exposed organisms (mg/kg).

- **Critical body residue (CBR):** The concentration in the organism (e.g., whole body, liver, or egg) that is associated with an adverse effect (mg COC/kg ww tissue).

- **EL-effect:** The effect level that results in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg-d) or concentration (mg/kg).

- **EL-no effect:** The effect level that does not result in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg-d) or concentration (mg/kg).

Most USEPA risk assessments typically estimate risk by comparing the exposure of receptors of concern to TRVs that are based on NOAELs. TRVs for this BERA were developed on the basis of both NOAELs and LOAELs to provide perspective on the range of potential effects relative to measured or modeled exposures.

Differences in the feeding behavior of aquatic and terrestrial organisms determine the type of toxicity endpoints that are most easily measured and most useful in assessing risk. For example, the dose consumed in food is more easily measured for terrestrial animals than for aquatic organisms since uneaten food can be difficult to collect and quantify in an aqueous environment. Therefore, for aquatic organisms, toxicity endpoints are more often expressed as concentrations in external media (e.g., water) or as accumulated concentrations in the tissue of the exposed organism (also called a "body burden"). In some studies, doses are administered via gavage, intraperitoneal injection into an adult, or injection into a fish or bird egg.

Where appropriate studies are available, TRVs for this BERA were selected on the basis of the most likely route of exposure, which for fish are expressed as CBRs (e.g., mg/kg whole body weight and mg/kg lipid in eggs or whole body) and for wildlife receptors (i.e., birds and mammals) are expressed as daily dietary doses (e.g., mg/kg whole body weight-d).

9.3.1.1 Methodology Used to Derive Toxicity Reference Values

The literature on toxic effects of COCs to animals includes studies conducted solely in the laboratory, as well as studies including a field component. Each type of study has advantages and disadvantages for the purpose of deriving TRVs for a risk assessment. For example, a controlled laboratory study can be designed to test the effect of a contaminant on the test species in the absence of the effects of other co-occurring contaminants. This is an advantage, since greater confidence can be placed in the conclusion that observed effects are related to exposure to the test compound. However, laboratory studies are often conducted on species that are easily maintained in the laboratory, rather than on wildlife species. Therefore, laboratory studies may have the disadvantage of being conducted on species that are less closely related to a particular receptor. This not a great disadvantage to the risk assessment, since the assessment endpoints evaluate feeding groups, as represented by individual receptor models. Field studies have the advantage that organisms are exposed to a more realistic mixture of contaminants than, for example, laboratory tests that expose organisms to a specific form of a contaminant (e.g., methylmercury chloride or Aroclor 1254). Field studies have the disadvantage that organisms are usually exposed to other contaminants, and observed effects may not be attributable solely to exposure to a specific contaminant.

Field studies were used in this BERA when they were available for species in the same taxonomic family as the receptor of concern and examined relevant sensitive endpoints, such as reproductive effects. When appropriate field studies were not available for a test species in the same taxonomic family as the receptor species of concern, laboratory studies or field studies on less-closely-related species were used to establish TRVs for the receptor species. The general methodology described in the following paragraphs was used to derive TRVs.

When appropriate chronic-exposure toxicity studies on the effects of contaminants on lethality, growth, or reproduction were not available for the species examined for a particular assessment endpoint, studies on other species were used to develop TRVs. In general, few receptor-specific studies were available. Therefore, avian TRVs were developed for application to avian receptors and mammalian TRVs were developed for application to mammalian receptors.

The general methodology used to develop LOAEL and NOAEL TRVs for this BERA is described below:

- If an appropriate LOAEL was unavailable for a phylogenetically similar species (e.g. within the same taxonomic family), the assessment used a study conducted on another species, preferably one that was closely related to the receptor of concern. Whenever several studies were available, professional judgment was used to select the most appropriate LOAEL. Interspecies uncertainty factors, which account for potential differences in sensitivity between a test species and a receptor, were not used in the development of the final TRVs for the risk assessment.

- In the absence of an appropriate NOAEL, an appropriate LOAEL may be divided by a conversion factor of 10 to estimate a NOAEL. The LOAEL to NOAEL conversion is similar to USEPA's derivation of human health reference dose (RfD) values, where LOAEL studies are adjusted by a factor of 10 to estimate NOAEL values (Dourson and Stara, 1983).

- When calculating chronic dietary dose-based TRVs (e.g., mg/kg-d) from data for sub-chronic tests, the sub-chronic LOAEL or NOAEL values were divided by a conversion factor of 10 to estimate chronic TRVs. The use of a conversion factor of 10 is consistent with the methodology used to derive human health RfDs (Dourson and Stara, 1983).

  USEPA has not established a definitive line between sub-chronic and chronic exposures for ecological receptors. This BERA generally follows Sample et al. (1996), which considers 10 weeks to be the minimum time for chronic exposure of birds and one year for chronic exposure of mammals (based on half the life span of laboratory rodents). However, in addition to duration of exposure, the time when contaminant exposure occurs is critical. Reproduction is a particularly sensitive life stage, due to the stressed condition of the adults and the rapid growth and differentiation occurring within the embryo (Sample et al., 1996). For many species, contaminant exposure of a few days to as little as a few hours during gestation and embryo development may produce severe adverse effects. Because TRVs were selected to evaluate the potential for adverse effects on wildlife populations and impaired reproduction is likely to affect populations, contaminant exposures of less than one year or 10 weeks, but that occur during reproduction, were considered to represent chronic exposures.

- In cases where TRVs were available as a dietary concentration (e.g., mg contaminant per kg food), a daily dose for birds or mammals was calculated on the basis of standard estimates of food intake rates and body weights (e.g., Sample et al., 1996; USEPA, 1993b).

9.3.2 Fish

Eleven COCs, i.e., antimony, arsenic, chromium, mercury/methylmercury, selenium, vanadium, zinc, DDT and metabolites, endrin, total PCBs, and dioxins/furans, were selected for fish. Risk was characterized based on measured body burdens in whole fish, which were then compared to body burden-based TRVs. Due to the limited range of body burden studies available, one set of TRVs was selected to apply to all fish species (Table 9-15).

9.3.2.1 Antimony

Antimony is a naturally occurring metal that is used in various manufacturing processes. Acute oral exposure of humans and animals to high doses of antimony or antimony-containing compounds may cause gastrointestinal disorders (e.g., vomiting, diarrhea), respiratory difficulties, and death at extremely high doses (Young, 1992). Subchronic and chronic oral exposure may affect hematologic parameters.

Doe et al. (1987) examined the toxicity of antimony to rainbow trout (*Oncorhynchus mykis*) in a 30-day test. Trout fingerlings (1.2 g) were exposed to antimony potassium tartrate in water at concentrations of 8 and 16 mg/L over a 30-day period. Fingerlings exposed to the higher dosage showed a reduction in survival of 50 percent, while those at the lower dosage showed no survival effects. Tissue residues were 9.0 mg/kg ww for fingerlings that showed reduced survival and 5.0 mg/kg ww for those with no survival effects. These values were selected for a LOAEL and NOAEL of 9.0 and 5.0 mg/kg ww, respectively. Sublethal antimony effect levels, such as reproductive endpoints, are likely to be much lower than these values.

9.3.2.2 Arsenic

Arsenic occurs naturally as sulfides and as complex sulfides of iron, nickel, and cobalt. However, anthropogenic input exceeds the amount of arsenic occurring naturally by about a factor of three (Eisler, 1988a). Inorganic forms of arsenic are more toxic than organic forms, and trivalent are more toxic than pentavalent forms. Arsenic toxicity varies between species and the effects can be altered by physical, chemical, and biological conditions. Health effects may occur to the respiratory, gastrointestinal, cardiovascular, and hematopoietic systems, and may range from reversible effects to cancer and death.

Gilderhus (1966, as cited in ERED) exposed bluegills to weekly applications of sodium arsenite herbicide in an artificial pond. Arsenite, being trivalent, is considered to be more potent than the pentavalent congener. At tissue concentrations of 1.7 mg/kg, abnormal ovary and oocyte development were observed, and at tissue concentrations of 2.2 to 11.6 mg/kg, decreased weight gains were observed. The effect of the abnormal ovary and oocyte development and decreased weight gain on growth and reproduction was unclear.

Diminished growth and survival was reported in immature bluegills when total arsenic residues in muscle is more than 1.3 mg/kg ww, and more than 5 mg/kg ww in adult bluegills (National Research Council Canada [NRCC], 1978). Therefore, 1.3 mg/kg ww was selected as a LOAEL to protect sensitive life stages. Walsh et al. (1977) determined that whole-body arsenic concentrations above 0.5 mg/kg may be harmful to fish. Therefore, a NOAEL of 0.5 mg/kg was selected for arsenic.

9.3.2.3 Chromium

Chromium plays a role in glucose and cholesterol metabolism and is thus essential to humans and animals. However, animals given lethal doses of various chromium compounds have exhibited symptoms including hypoactivity, lacrimation, mydriasis, diarrhea, changes in body weight, pulmonary congestion, fluid in the stomach and intestine, erosion and discoloration of the gastrointestinal mucosa, diarrhea, and gastric ulcers (Daugherty, 1992).

Chromium toxicity was evaluated based on a study by Van der Putte et al. (1981). Rainbow trout (*Oncorhynchus mykis*) were exposed to hexavalent chromium in water concentrations ranging from 2 to 50 mg/L over a period of four days. Significant lethality was noted at exposure concentrations corresponding to body burdens greater than 7.8 mg/kg ww. No mortality was noted at exposures corresponding to 2.3 mg/kg ww. Because of the short duration of the study, an uncertainty factor of 0.1 was used to extrapolate from subchronic to chronic exposure, resulting in a chromium NOAEL of 0.23 mg/kg ww and a LOAEL of 0.78 mg/kg ww. Sublethal chromium effect levels, such as reproductive endpoints, are likely to be much lower than these values.

9.3.2.4 Mercury/Methylmercury

Methylmercury is the most hazardous mercury species, due to its high lipid solubility and ionic properties that allow it to penetrate the membranes of living organisms. Methylmercury adversely affects reproduction, growth, behavior, osmoregulation, and oxygen exchange in aquatic organisms. Most mercury in fish is methylmercury, as confirmed by the data from Onondaga Lake. Therefore, all mercury concentrations in fish were considered to be methylmercury. Methylmercury readily penetrates the blood-brain barrier, produces brain lesions, spinal cord degeneration, and central nervous system dysfunctions.

There is both field and laboratory evidence that diet is the most important route of fish exposure to methylmercury, as it contributes 90 percent or more of the methylmercury accumulated. The assimilation efficiency for uptake of dietary methylmercury in fish is approximately 65 to 80 percent or greater.

Reproductive endpoints are generally more sensitive than growth or survival endpoints, with embryos and the early developmental stages being the most sensitive. Mercury can be transferred from tissues of the adult female to the developing embryo. Sublethal and lethal effects on fish embryos are associated with mercury residues in eggs that are perhaps 1 to 10 percent of the residues associated with toxicity in adult fish. Mercury concentrations in intoxicated rainbow trout range between 4 and 30 mg/kg (whole body), while intoxicated embryos contain 0.07 to 0.1 mg/kg (Weiner and Spry, 1996).

The toxic concentration of mercury compounds can vary by an order-of-magnitude or more, depending on the exposure condition. For example, toxicity is greater at elevated temperatures (Armstrong, 1979) and at lower oxygen content (Sloof et al., 1991).

The effects on aquatic organisms due to interactions of mercury with cadmium, copper, selenium, and zinc were found to be dependent on exposure concentrations (Birge et al., 1979). The interaction of mercury and other trace elements (e.g., selenium and zinc) can be both less than additive (antagonistic) and greater than additive (synergistic), depending primarily on exposure concentrations and the form of mercury. Effects were generally antagonistic at lower exposure levels and synergistic at higher levels. Exposure to low concentrations of mercury may not result in mortality directly, but may retard growth, thereby increasing the risk of predation (NOAA, 1996).

No standards that would be protective of aquatic organisms have been established for mercury concentrations in fish tissue. The current Food and Drug Administration (FDA) action level for the protection of human health, based only on methylmercury in the edible flesh of fish and shellfish, is 1 mg/kg (USFDA, 1984).

Friedmann et al. (1996) studied concentrations frequently observed in North American lakes to investigate the effects of dietary methylmercury on growth, gonadal development, and plasma cortisol levels in juvenile walleye (*Stizostedion vitreum*) over a six-month period. Reduced testicular development and immune function were observed at whole-body concentrations of 0.25 mg/kg ww. Rainbow trout exposed to mercuric chloride for 400 to 528 days showed significant reduction in alevin survival (four-day post-hatch) and a significant increase in teratogenic effects at a concentration of 0.5 mg/kg ww in ovary tissue (Friedmann et al., 1996).

NOAA (2002) has summarized toxicity associated with mercury in tissues. Based on their review of peer-reviewed studies, NOAA developed a mercury NOAEL and LOAEL of 0.1 and 0.3 mg/kg ww, respectively, for use at the LCP National Priorities List (NPL) EPA Region 4 site in Brunswick, Georgia (Mehran, 2002, pers. comm.). These TRVs were also selected for use in this BERA.

9.3.2.5 Selenium

Selenium is beneficial or essential in amounts from trace to part-per-billion (ppb) concentrations for humans and some plants and animals, but can be toxic at higher concentrations. Selenium chemistry is complex, and its metabolism and degradation are significantly modified by interaction with such elements as heavy metals, agricultural chemicals, microorganisms, and a variety of physicochemical factors (Eisler, 1985).

Fish with high body burdens of selenium failed to reproduce and exhibited teratogenic deformities in Belews Lake, North Carolina, which received dissolved selenium in wastewater from a coal-fired electricity generating facility (Lemly, 1997). Low waterborne concentrations of selenium eliminated 16 of 20 fish species present in the lake, and rendered the adults of two species sterile (Cumbie and Van Horn, 1978; Lemly, 1985). In these fish, selenium levels were elevated in liver (up to 21.4 mg/kg ww) and other tissues; kidney, heart, liver, and gills showed altered histopathology; and there were changes in blood chemistry. The ovaries of fish from Belews Lake had numerous necrotic and ruptured egg follicles that may have contributed to the population extinctions (Sorensen et al., 1984).

A survey performed ten years after the selenium releases to Belews Lake were stopped found developmental abnormalities in young fish, indicating that selenium-induced teratogenesis and reproductive impairment were occurring, and that concentrations of selenium in benthic food organisms are sufficient to cause mortality in young bluegill and other centrarchids due to Winter Stress Syndrome (WSS) (Lemly, 1997). WSS occurs when sublethal effects (metabolic stress) due to selenium are present at the same time as the arrival of cold water temperatures in late autumn. Cold weather and the associated short photoperiod of winter programs the fish for reduced activity and food intake, and they do not respond to the metabolic stress with increased feeding. If exposure to selenium persists, stored body fat necessary for overwintering is used up, fitness drops, and death may result.

Another, more compelling, effect of selenium on fish in regard to reproduction was also noted by Lemly (1997). Absorption of selenium passed from parents to their offspring in eggs causes morphological abnormalities as the young develop, if the concentrations in eggs reach 15 to 20 mg/kg dw (Gillespie and Baumann, 1986; Woock et al., 1987; Coyle et al., 1993). Using a conversion factor of fish whole-body values multiplied by 3.3 to calculate egg concentrations, based on the work of Lemly (1996, 1997), the threshold whole-body concentrations are between 4.5 and 6.1 mg/kg dw. Given an average percent solids of about 24 percent in Onondaga Lake fish, this translates to roughly 1.1 to 1.5 mg/kg ww.[1] The lower end of this range (1.1. mg/kg ww) was selected as the LOAEL, and an uncertainty factor of 0.1 was applied to the LOAEL to derive a NOAEL of 0.11 mg/kg ww (0.45 mg/kg dw).

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[1] Each individual fish sample analyzed as wet weight was converted to dry weight based on its percent solids.

9.3.2.6 Vanadium

Vanadium is a natural constituent of sediment and water, as well as being found in fuel oils and coal. In water, vanadium can exist in both soluble forms and as a precipitate. Vanadium from water can be taken up and accumulated by fish.

Hilton and Bettger (1988) studied the effects of vanadium on juvenile rainbow trout. Trout were exposed to concentrations of 10.2 and 1.2 mg/kg sodium orthovanadate in their diet. They found reduced feeding and body weight and digestive tract distress at a concentration of 0.41 mg/kg ww in fish carcasses. They also noted protruding abdomens and darkened skin coloration on these fish. Trout that received a dose of 1.2 mg/kg in their diet had carcass concentrations of 0.02 mg/kg ww and showed no effects. Based on this study, a LOAEL of 0.41 mg/kg ww was selected. As the NOAEL concentration was about 20 times lower than LOAEL concentration, an uncertainty factor of 0.1 was applied to the LOAEL to derive a NOAEL of 0.041 mg/kg ww to avoid an overly conservative estimate of risk.

9.3.2.7 Zinc

Zinc is present in the environment naturally, but high concentrations come from activities such as mining, steel production, coal burning, and burning of waste. The toxicity of zinc to aquatic organisms is influenced by many factors, such as the temperature, hardness, and pH of the water, and previous zinc exposure of the organisms. Several fish kills in recent years have been attributed to zinc from runoff and discharges from mining areas and smelters. However, the concentrations causing mortality were generally not well documented, and, in many cases, high levels of other metals were also present.

A study on the effects of zinc on the American flagfish (*Jordanella floridae*) by Spehar (1976) was used to derive zinc TRVs. In this study, flagfish were exposed to zinc sulfate ($ZnSO_4$) in water at concentrations ranging from 26 to 139 μg/L. Reduced growth of females was seen at a dose of 51 μg/L and was the most sensitive measure of zinc toxicity. No effects were seen at exposure to 26 μg/L. These exposures translated into tissue concentrations of 40 and 34 mg/kg ww, respectively. Therefore, 40 mg/kg ww was selected as a LOAEL and 34 mg/kg ww was selected as the NOAEL.

9.3.2.8 DDT and Metabolites

Dichlorodiphenyl trichloroethane (DDT) was used as a pesticide until it was banned in 1972 due to unacceptable risks to the environment and potential harm to human health. DDT was developed as the first of the modern insecticides early in World War II. It was initially used with great effect to combat malaria, typhus, and the other insect-borne human diseases among both military and civilian populations. DDT came into wide agricultural and commercial usage in this country in the late 1940s.

DDT is toxic to several fish species, with the greatest mortalities in the younger age groups. DDT-contaminated feed has caused massive mortalities of sac fry of brook, rainbow, and cutthroat (*Oncorhynchus clarki*) trout in hatcheries (Connell and Miller, 1984). Rainbow trout and coho salmon

(*Oncorhynchus kisutch*) have been similarly affected in DDT-contaminated lakes (Connell and Miller, 1984). The organochlorines accumulate in eggs and can lead to the death of fry as the yolk sac is absorbed (Connell and Miller, 1984).

The toxicity to fish of DDT and its metabolites was based on a reproductive study of brook trout by Macek (1968). In this investigation, yearling trout were exposed to DDT through their diets at three dose levels (0.5, 1, and 2 mg/kg-week) for 156 days, including five months prior to spawning, with fertilized eggs produced from the control (i.e., no DDT exposure) and 1 and 2 mg/kg-week doses. A significant reduction in mature egg production was noted at the highest dose level. Increased mortality in eggs and sac fry were significantly higher in all mating combinations that received either one or both gametes from a treated parent. Observations indicated that mortality of fry may be due to DDT being released from the yolk fat (i.e., fry feeding) during the period of its maximum utilization (15th week). Total residues in adults corresponded to the levels of exposure.

The 1 mg/kg-week dose was selected as the LOAEL, based upon fry mortality. The mean body burden of DDT and metabolites (DDE and DDD) of fish treated with 1 mg/kg-week at the end of the exposure period was 2.9 mg/kg ww. The mean concentration of DDT and metabolites in the control group was 0.6 mg/kg ww, which was selected as a NOAEL.

9.3.2.9 Dioxins/Furans

Dioxins and furans are byproducts of chemical manufacturing, the result of incomplete combustion of materials containing chlorine atoms and organic compounds, or formed during the disinfection of complex effluents (e.g., pulp and paper effluents) containing many organic constituents. These substances have been associated with a wide variety of toxic effects in animals, including acute toxicity, enzyme activation, tissue damage, developmental abnormalities, and cancer.

To assess toxicity, chlorinated dioxins and furans are classified at varying levels of potency of 2,3,7,8-TCDD (Eastern Research Group [ERG], 1998). Dioxin/furan toxicity to fish was evaluated using toxicity equivalents [TEQs] for fish taken from Van den Berg et al. (1998).

Laboratory and field studies on the effects of dioxin-like compounds TEQs on fish typically report concentrations of TEQs in fish eggs, rather than in the whole body, since eggs represent a more sensitive life stage. Comparison of effect levels, such as NOAELs or LOAELs, reported as wet weight concentrations in eggs to whole-body tissue concentrations in adult Onondaga Lake fish is complicated by the fact that eggs and whole-body adult fish tend to have different lipid contents and concentrations of lipophilic contaminants, such as TEQs.

However, if TEQs are assumed to partition equally into the lipid phase of the egg and into the lipids in the tissue of adult fish (Niimi, 1983), then lipid-normalized concentrations in fish eggs that are associated with adverse effects (μg TEQs/kg lipid in egg) can be compared to lipid-normalized tissue concentrations of TEQs in adult fish (μg TEQs/kg lipid in whole-body adult). Therefore, LOAEL and NOAEL TRVs were established for TEQs in fish on a lipid-normalized basis, so that measured whole-body concentrations of TEQs in fish can be compared to TRVs established from studies on fish eggs.

A study on lake trout by Walker et al. (1994) was selected for the dioxin/furan TRVs. In this study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 μg TEQs/kg lipid. This effect was not observed at a concentration of 0.29 μg TEQs/kg lipid. These results were similar to other studies performed by Walker et al. (1992) and Walker and Peterson (1994). The values from this study were selected as TRVs in this assessment for a LOAEL of 0.6 μg TEQs/kg lipid and a NOAEL of 0.29 μg TEQs/kg lipid. Because the experimental study was based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor was not applied.

9.3.2.10 Endrin

Endrin is a rodenticide used to control mice and voles, and an insecticide used on cotton, rice, and maize. Closely related to aldrin and dieldrin, endrin is the most toxic of the three in the aquatic environment (UNEP, 2002) and its metabolites are more toxic than endrin itself.

Jarvinen and Tyo (1978) studied the effects of chronic exposure of fathead minnows (*Pimephales promelas*) to endrin concentrations in the water or food (clams), or both, for 300 days encompassing reproduction. Tissue residues were analyzed at present intervals for first-generation fish, and were also determined for embryos, larvae at hatch, and 30-day progeny. Endrin in the food (0.63 ppm) significantly reduced survival of the fathead minnows, and fish exposed to both endrin sources had lower survival than those exposed to either source alone.

Endrin residues in embryos and larvae were highest and larval survival lowest for progeny of adults exposed to endrin in both food and water. Survival of 30-day progeny was significantly reduced at all test exposures (0.63 ppm in the food, water exposures of 0.14 and 0.25 ppb, and all combinations of food and water exposure). Reduced survival was observed in larvae with a tissue residue of 0.24 mg/kg ww. This value was selected as a LOAEL and a factor of 0.1 was used to derive a NOAEL for LOAEL and NOAEL values of 0.24 and 0.024 mg/kg ww, respectively.

9.3.2.11 Polychlorinated Biphenyls

PCBs are industrial compounds that were used in a broad range of commercial applications until their manufacture was banned in 1976 under the Toxic Substances Control Act (TSCA) (15 U.S.C. Sec. 2601 et seq.). The toxicity of PCBs has been shown to manifest itself in many different ways, among various species of animals. Typical responses to PCB exposure in animals include wasting syndrome, hepatotoxicity, immunotoxicity, neurotoxicity, reproductive and developmental effects, gastrointestinal effects, respiratory effects, dermal toxicity, and mutagenic and carcinogenic effects. Some of these effects are manifested through endocrine disruption. PCB exposure through diet and water have been reported to cause a number of deleterious effects in fish survival, growth, egg production, and hatching success, as well as survival and development of progeny (Defoe et al., 1978; Cleland et al., 1988; Fisher et al., 1994).

A study using the sheepshead minnow (*Cyprinondon variegatus*) by Hansen et al. (1974) was selected as the most appropriate study to derive PCBs TRVs. This study established a NOAEL of 1.9 mg PCBs/kg and a LOAEL of 9.3 mg PCBs/kg for the sheepshead minnow. This study was based on a flow-through bioassay of Aroclor 1254 on adult female fish. Fish were exposed for 28 days, and then egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality.

Survival of fry to one week of age was 77 percent for eggs from adults from the 0.32 μg/L concentration in water treatment (average 9.3 mg/kg in tissue of females), as compared to 95 percent survival of fry from control adults and 97 percent survival of fry from adults from the NOAEL treatment (0.1 μg/L; average 1.9 mg/kg in tissue of females). A LOAEL of 9.3 mg/kg in tissue and a NOAEL of 1.9 mg/kg in tissue were selected for this BERA. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (e.g., food, water, or sediment, or injected dose), a subchronic-to-chronic conversion factor was not applied.

9.3.3 Amphibians and Reptiles

A worldwide decline in amphibian and reptile populations has caused great concern in the scientific community (Environment Canada, 2001). As environmental contaminants have been implicated as a possible cause of some declines, there has been a substantial increase in the amount of amphibian and reptile ecotoxicology research conducted over the last decade. Amphibians may be exposed to toxic compounds through several routes because of their semipermeable skin, the development of their eggs and gill-breathing larvae in the water, and their changing position in the food web from herbivorous tadpoles to carnivorous adults (Gutleb et al., 1999). Amphibians typically have both terrestrial and aquatic life stages during which they may be susceptible to the effects of environmental contaminants. In addition, amphibians are important food organisms for a large variety of fish, birds, and mammals, and can be of major ecological significance (Nebeker et al., 1995). Reptiles are long-lived, sedentary beings and therefore may be good "biomonitors" of their local environment.

Effects from contaminant exposure may vary depending on exposure route, the point in time of exposure during the life cycle, and the length and intensity of the exposure. The embryo is generally the most sensitive life stage (Pèrez-Coll and Herkovits, 1996), so high concentrations of contaminants/stressors during embryonic stages (generally spring) may have significant repercussions on amphibian populations.

Although herpetofaunal toxicity studies are increasing (e.g, see Environment Canada's Reptile and Amphibian Toxicity Literature database [Environment Canada, 2001] or California EPA's Exposure Factor and Toxicity database [CAL/EPA, 2002]), there are still many data gaps. Therefore, general water and sediment quality criteria values for aquatic organisms are the best available values for many compounds. Based on the limited herpetological toxicity data, a quantitative analysis of risk to amphibians and reptiles is not performed in this BERA. It is acknowledged, however, that contaminants may adversely affect herpetofauna. For example, Zoll et al. (1988) examined the genotoxicity and bioaccumulation of mercury in newts. They observed broken chromosomes and chromosome aberrations in blood smears from larvae exposed to mercuric chloride and methylmercuric chloride. Bioaccumulation ratios after 12 days were 600 for mercuric chloride and 1,200 for methylmercuric chloride. Metals, such as cadmium, may also be bioaccumulated. Larval salamanders exposed to cadmium in the water had tissue concentrations up to 63 times the water concentration and exhibited adverse growth effects (Nebeker et al., 1995).

9.3.4 Birds and Mammals

Twenty-eight COCs were selected to evaluate potential risk to wildlife receptors (see Table 6-2). This section discusses the toxicity of selected COCs and the development of the specific baseline assessment TRVs necessary to characterize risk for terrestrial vertebrates. Generally, reproductive endpoints were preferred for development of TRVs. In instances where no reproductive studies were available, or where studies with other endpoints were considered to be superior, based on professional judgment, non-reproductive endpoints were chosen. TRVs for non-reproductive endpoints were generally higher than for reproductive endpoints. Tables 9-16 and 9-17 summarize the TRVs selected for terrestrial wildlife.

The model used to assess the potential risks was based on a numerical comparison of the modeled exposure rate over the TRV to derive the hazard quotient (HQ), as follows:

$$HQ = \frac{EER}{TRV}$$

where:

HQ	=	hazard quotient or the ratio of the exposure and the TRV (unitless)
EER	=	estimated exposure rate determined at the mean and 95 percent UCL of the mean COC concentrations in Onondaga Lake (mg/kg body weight [bw] per day)
TRV	=	toxicity reference value for the no effects or lowest observed effects thresholds (mg/kg body weight per day)

Exposure rates were evaluated using the NOAEL and LOAEL to provide a range of ecological risk. Considerations of uncertainty in the TRV predictions are discussed in Chapter 11. The derivations of specific TRVs are described below.

9.3.4.1 Arsenic

Arsenic is a naturally occurring element that is used as a wood preservative and also in insecticides and herbicides. Effects of arsenic exposure in mammals include stomach upset and diarrhea. Large doses may cause low birth weight, fetal malformations, or death.

The toxicity of inorganic compounds containing arsenic depends on the valence or oxidation state of the arsenic as well as on the physical and chemical properties of the compound in which it occurs (Sample et al., 1996). Trivalent (As^{+3}) compounds such as arsenic trioxide (As_2O_3), arsenic trisulfide (As_2S_3), and sodium arsenite ($NaAsO_2$), are generally more toxic than pentavalent (As^{+5}) compounds such as arsenic pentoxide (As_2O_5), sodium arsenate (Na_2HAsO_4), and calcium arsenate [$Ca_3(AsO_4)_2$]. The relative toxicity of the trivalent and pentavalent forms may also be affected by factors such as water solubility; the more toxic compounds are generally more water soluble. This BERA evaluates the effects of the trivalent form of arsenic.

Arsenic was selected as a COC for one avian receptor, the tree swallow (*Tachycineta bicolor*), and four mammalian receptors the little brown bat (*Myotis lucifugus*), short-tailed shrew (*Blarina brevicauda*), mink (*Mustela vison*), and river otter (*Lutra canadensis*).

The avian TRVs for arsenic were based on a study by US Fish and Wildlife Service (USFWS, 1969) where copper acetoarsenite (44 percent As^{+3}) was fed to cowbirds (*Molothrus ater*) at four dose levels. Cowbirds at the two highest dose levels (675 and 225 ppm) experienced 100 percent mortality, while those in the two lower groups (75 and 25 ppm) experienced 20 percent and 0 percent mortality, respectively. Because the study considered exposure over seven months, the 75 ppm Paris green (33 mg/kg As^{+3}) and the 25 ppm Paris green (11 mg/kg As^{+3}) doses, equivalent to 7.38 mg/kg-d and 2.46 mg/kg-d, were considered to be chronic LOAELs and NOAELs, respectively.

The mammalian TRVs for arsenic were developed based on a study by Schroeder and Mitchener (1971). Mice were exposed to 5 ppm arsenite in drinking water over three generations. This concentration was associated with a decrease in litter size and is, therefore, considered a potential population level LOAEL. An increase in the male-to-female ratio of offspring was also observed. Assuming a drinking water intake rate of 0.0075 L/d for a 30 g mouse, a LOAEL of 1.26 mg/kg-day was derived. An uncertainty factor of 0.1 was applied to derive a NOAEL of 0.126 mg/kg-day.

9.3.4.2 Barium

Barium is a naturally occurring element common in carbonate-based soils and metamorphic parent materials. Barium is used industrially in the production of paints, bricks, tiles, glass, and rubber. Exposure to barium can cause high blood pressure, changes in the function and chemistry of the heart, decreased life span, and decreased body weight.

Barium was selected as a COC for two avian receptors, the tree swallow and mallard (*Anas platyrhynchos*), and two mammalian receptors, the little brown bat and short-tailed shrew.

The avian TRVs were based on a study by Johnson et al. (1960) where one-day old chicks were fed barium throughout the 4 week study period. While barium exposures up to 2,000 ppm produced no mortality, chicks in the 4,000 to 32,000 ppm groups experienced 5 percent to 100 percent mortality. Because 2,000 ppm was the highest nonlethal dose, this dose was considered to be a subchronic NOAEL.

The 4,000 ppm dose was considered to be a subchronic LOAEL. Chronic NOAELs and LOAELs were estimated by multiplying the subchronic NOAELs and LOAELs by a subchronic to chronic uncertainty factor of 0.1 for a body weight-normalized NOAEL of 20.8 mg/kg-d and a LOAEL of 41.7 mg/kg-d.

Toxicity and carcinogenicity studies of barium chloride dihydrate were conducted by administering the chemical to rats and mice in drinking water for 13 weeks and for two years (National Toxicology Program [NTP], 1994). In the chronic study, male and female rats (60 animals/dose group/sex) received drinking water containing 0, 500, 1,250, or 2,500 mg/L barium chloride dihydrate (equivalent to a dose of 0, 15, 45 and 75 mg/kg-day) for 104 weeks (males) or for 105 weeks (females).

Increased relative kidney weight was seen in the females at 2,500 ppm, indicating that it may be a chronic NOAEL or LOAEL for rats. When considered together with the results in the 13-week subchronic NTP (1994) study in rats, in which increased relative and absolute kidney weights were seen in female rats receiving 2,000 ppm barium in drinking water (115 mg Ba/kg-day) and kidney lesions at 4,000 ppm (180 mg Ba/kg-day), greater increases in relative and absolute kidney weights were seen in female rats. Increased relative kidney weight in females of the two-year study are suggestive of potential renal effects. Therefore, 75 mg Ba/kg-day was selected as a chronic LOAEL and 45 mg Ba/kg-day as the chronic NOAEL for mammals, based on renal effects (USEPA, 1999b).

9.3.4.3 Cadmium

Cadmium is used to manufacture batteries, pigments, metal coatings, and plastics. Inhalation of cadmium is carcinogenic, and rats have been shown to develop lung cancer after exposure (USEPA, 1987a). Ingestion can cause high blood pressure, iron-poor blood, liver disease, and nerve or brain damage. It has also been demonstrated that rats have fewer litters, and pups may have more birth defects than usual when exposed to cadmium orally (Sutou et al., 1980).

Cadmium was selected as a COC for two avian receptors, the mallard duck and tree swallow, and two mammalian receptors, the little brown bat and short-tailed shrew.

The avian TRVs were derived using a study by White and Finley (1978). Mallard ducks were exposed to 1.6, 15.2, and 210 ppm cadmium chloride. Mallards in the 210 ppm group produced significantly fewer eggs than those in the other groups. Because the study considered exposure over a period of 90 days, the 15.2 ppm cadmium dose was considered to be a chronic NOAEL and the 210 ppm dose was considered to be a chronic LOAEL. Adjusted for mallard body weight, these equal a NOAEL of 1.45 mg/kg-day and a LOAEL of 20 mg/kg-day.

For the mammalian TRV, a study by Sutou et al. (1980), in which rats were exposed to cadmium (as $CdCl_2$) at four dose levels (0, 0.1, 1, and 10 mg/kg-day) by oral gavage, through mating and gestation (six weeks), was selected. Adverse reproductive effects, including reduced fetal implantations, reduced fetal survivorship, and increased fetal resorptions were observed in the rats exposed to 10 mg/kg-day. Numbers of total implants and live fetuses in the 1 mg/kg-day decreased slightly, but there was no significant difference from the control. As the study was conducted during reproduction, exposures were considered chronic even though exposure lasted only six weeks. Therefore, 1 mg/kg-day dose was considered to the NOAEL and a dose of 10 mg/kg-day was considered the LOAEL TRV for the evaluation of risk to mammals.

9.3.4.4 Chromium

Chromium is a naturally occurring element found in rocks, animals, plants, and soil. Chromium compounds are used for chrome plating, the manufacture of dyes and pigments, leather tanning, and wood preserving. The metal chromium is used to make steel and other alloys. Inhalation of high levels of chromium may cause lung cancer. Ingesting large amounts may result in the development of skin ulcers, stomach upsets, and kidney and liver damage.

Chromium was selected as a COC for all five avian receptors (belted kingfisher [*Ceryle alcyon*], great blue heron [*Ardea herodias*], osprey [*Pandion haliaetus*], mallard, and tree swallow) and all four mammalian receptors (little brown bat, short-tailed shrew, mink and river otter).

To derive the avian TRVs, a study by Haseltine et al. (unpublished data, cited in Sample et al., 1996) was selected. Black ducks (*Anas rubripes*) were exposed to chromium(III) (as $CrK[SO_4]_2$) at two dose levels (10 and 50 ppm in food) for 10 months through reproduction. Duckling survival was reduced at the 50 ppm dose level, while no significant differences were observed at the 10 ppm dose level. Because the study considered exposure throughout a critical life stage (reproduction), the 50 ppm dose was considered to be a chronic LOAEL, and the 10 ppm dose was considered to be a chronic NOAEL. Assuming that the body weight of a mallard is 1.25 kg (Dunning, 1993) and the food consumption rate is 10 percent (Heinz et al., 1989) the NOAEL was determined to be 1 mg/kg-day and the LOAEL TRV was determined to be 5 mg/kg-day.

A study by MacKenzie et al. (1958) was used to derive the mammalian NOAEL. Rats were exposed to chromium(VI) (as $K_2Cr_2O_4$) at six dose levels in drinking water (0.45, 2.2, 4.5, 7.7, 11.2, and 25 ppm in water) for one year. Because no adverse effects were observed at any of the dose levels, the maximum dose (25 ppm chromium in water, or 3.28 mg/kg-day) was considered to be a chronic NOAEL. The assumptions used in TRV calculations included a body weight of 0.35 kg and water consumption rate of 0.046 L/day for rats.

The LOAEL TRV for exposure to chromium was based on a study by Steven et al. (1976, as cited in Sample et al. 1996). Rats were exposed daily to 134 and 1,000 ppm chromium(VI) in drinking water for three months. Increases in mortality were noted at 1,000 ppm, which was considered to be a subchronic LOAEL. A chronic LOAEL was estimated by multiplying the subchronic LOAEL by a subchronic-chronic uncertainty factor of 0.1. Based on the same body mass and intake rates used to derive the NOAEL, a LOAEL TRV of 13.14 mg/kg-day was derived.

9.3.4.5 Copper

Copper is used as component of some insecticides and fungicides and may also enter the environment through industrial activities. It is a naturally occurring element, but at high doses, it may reduce growth and result in mortality.

Copper was selected as a COC for two avian receptors (mallard, and tree swallow) and one mammalian receptors (little brown bat).

A study by Mehring et al. (1960) examining the effects of copper oxide on chicks for 10 weeks was used to derive avian TRVs. Chicks were fed one of 11 dose levels in their diet ranging from 36.8 to 1,150 ppm. While consumption of copper up to 570 ppm had no effect of growth of chicks, 749 ppm copper in the diet reduced growth by over 30 percent and produced 15 percent mortality. Because this study was 10 weeks in duration, the 570 and 749 ppm Cu doses were considered to be a chronic NOAEL and LOAEL, respectively. These doses adjusted for body weight and intake translated into a NOAEL of 47 mg/kg-d and a LOAEL of 61.7 mg/kg-d (Sample et al., 1996).

A study by Aulerich et al. (1982) on young mink fed supplemental copper (copper sulfate) at concentrations of 25, 50, 100, and 200 ppm in their diet was used to derive the mammalian TRVs. A concentration of 60.5 mg/kg was present in the base feed. Consumption of copper at all but the lowest does level increased the percentage mortality of mink kits. Because this study was approximately one year in duration and considered exposure during reproduction, the 25 ppm supplemental copper (85.5 ppm total copper) dose was considered to be a chronic NOAEL, equivalent to 11.7 mg/kg-day based on body weight (1 kg) and intake rate (137 g/d), and the 50 ppm supplemental copper (110.5 ppm total copper) dose was considered to be a chronic LOAEL, equivalent to15.14 mg/kg-d.

9.3.4.6 Lead

Lead is a metal that ranges from 0.1 to 10 ppm in ultramafic rocks and calcareous sediments (Kabata-Pendias and Pendias, 1992). Lead is used industrially in the production of batteries, ammunition, ceramics, and medical and scientific equipment. The toxic effects of lead on aquatic and terrestrial organisms are extremely varied and include mortality, reduced growth and reproductive output, blood chemistry alterations, lesions, and behavioral changes. However, some of these effects exhibit general trends in their toxic mechanism. Generally, lead inhibits the formation of heme, adversely affects blood chemistry, and accumulates at hematopoietic organs (Eisler, 1988b). At high concentrations, near levels causing mortality, marked changes to the central nervous system occur prior to death (Eisler, 1988b).

Lead was selected as a COC for three avian receptors, the tree swallow, belted kingfisher, and red-tailed hawk, and two mammalian receptors, the little brown bat, and short-tailed shrew.

The avian TRV was developed from a study by Edens et al. (1976). Japanese quail (*Coturnix coturnix japonica*) were exposed to lead acetate in feed at four dose levels (1, 10, 100, and 1,000 ppm in food) for 12 weeks, through reproduction. The 10 ppm lead concentration (11 ppm dry weight [dw]) resulted in no significant adverse reproductive effects, but even as little as 1 ppm of lead caused a marked decline in egg production. Exposure at 100 ppm (110 ppm dw) resulted in a reduction in hatching success by 28 percent. Assuming a body weight of 0.15 kg (Vos et al., 1971) and food consumption rate of 16.9 g dw/kg body weight-day (based on Nagy, 1987), a NOAEL of 1.18 mg/kg-day and LOAEL of 11.8 mg/kg-day were calculated.

The mammalian TRVs for lead were based on a laboratory study by Azar et al. (1973) of three generations of rats given doses of 10, 50, 100, 1,000, and 2,000 ppm lead acetate in their food. Lead exposures of 1,000 and 2,000 ppm resulted in reduced offspring weights and produced kidney damage in the young. Therefore, the 100 ppm dose (8 mg/kg-day) was considered to be a chronic NOAEL and the 1,000 ppm dose (80 mg/kg-day) was considered to be a chronic LOAEL.

9.3.4.7 Manganese

Manganese is used as component of some insecticides and fungicides and may also enter the environment through industrial activities. It is a naturally occurring element, but at high doses, it has been shown to cause reproductive effects and stimulate tumors (NIOSH, 2002).

Manganese was selected as a COC for one mammalian receptor, the little brown bat.

A study by Lasky et al. (1982) where rats were fed manganese oxide (Mn_3O_4) in their diet at three dose levels (350, 1,050, and 3,500 mg/kg supplemented manganese + 50 mg/kg manganese in base diet) for 224 days. Pregnancy percentage and fertility among rats consuming 3,550 ppm manganese in their diet was significantly reduced No effects were observed at lower manganese exposure levels. Therefore the 1,100 ppm Mn dose was considered to be a chronic NOAEL and the 3,550 ppm Mn dose was considered to be a chronic LOAEL, equivalent to doses of 88 and 284 mg/kg-day, respectively.

9.3.4.8 Mercury (Inorganic)

Mercury exists in the environment in different chemical forms. The predominant species in water, soil, and sediment is ionic or inorganic mercury (Hg^{2+}). Ionic mercury can exist in a free ionic form (as chlorides or hydroxides), but most is adsorbed or chemically bound to clays, sulfides, and/or organic matter.

The kidney is the major reservoir of inorganic mercury in birds and mammals. In renal tissue, mercury binds to metallothionein. Consequently, the major toxic effect of inorganic mercury is kidney damage – specifically, necrosis of the proximal tubular cells. Inorganic mercury, unlike the metallic or organic species, is incapable of crossing the blood-brain barrier and, therefore, does not exhibit neurotoxicity. Other systemic effects include gastrointestinal damage and cardiovascular effects. There is limited evidence that inorganic mercury may pose some reproductive toxicity.

Inorganic mercury was selected as a COC for all avian and mammalian wildlife receptors.

The evaluation of risk to birds from exposure to inorganic mercury was based on a study by Hill and Shaffner (1976). Japanese quail chicks were fed 2, 4, 8, 16, or 32 ppm mercury as mercuric chloride until one year of age. There were no significant effects on food consumption, growth rate, or body weight maintenance at any dose. Hatchability and eggshell thickness were also unaffected at 4 ppm (4.4 ppm dw), but egg hatching rates were depressed by 16.1 percent at 8 ppm (8.8 ppm dw). Assuming a body weight of 0.15 kg (Vos et al., 1971) and an ingestion rate of 0.169 kg dw/kg body weight-day (derived from Nagy, 1987), a NOAEL TRV of 0.45 mg/kg-day and a LOAEL TRV of 0.90 mg/kg-day were calculated. The actual TRVs may be lower because the study did not discuss hatchling survival, which is influenced by some forms of mercury (e.g., Heinz, 1974).

The evaluation of risk to mammals from exposure to inorganic mercury was based on a study by Aulerich et al. (1974). Mink were orally dosed with 10 ppm (40 ppm dw) mercuric chloride for six months over gestation. No reproductive effects were observed. This was, therefore, considered to be the NOAEL for exposure of mammals to inorganic mercury. Assuming a body weight for penned mink of 1 kg (USEPA, 1993b) and a food consumption rate of 0.548 kg dw/kg body weight-day (based on the observations of Bleavins and Aulerich, 1981), the NOAEL TRV was determined as 1 mg/kg-day. Only one dose level was examined in the study, so the LOAEL TRV was calculated by applying an uncertainty factor of 10 for a LOAEL of 10 mg/kg-day.

9.3.4.9 Methylmercury

Methylmercury (CH_3Hg^+; $[CH_3]_2Hg$) represents a small, but significant, fraction of total mercury (approximately 10 percent) in the water column because of its high toxicity and natural tendency to bioaccumulate in upper trophic level prey. Most of the mercury in fish is present as methylmercury, providing exposure pathways for piscivorous and semi-piscivorous animals. Mercury methylation also occurs in wetlands (see Chapter 6, Section 6.3.1), providing exposure pathways for terrestrial wildlife.

Methylmercury was selected as a COC for all avian and mammalian wildlife receptors.

Methylmercury in birds has been demonstrated to affect various organ systems, with embryos being more sensitive than adults (Eisler, 1987a). Toxic effects of methylmercury include decreased reproductive success, altered behavior, hepatic lesions, ataxia, weakness, muscular atrophy, and death. Reproductive effects of mercury in birds include reduced hatchability (due to increases in early mortality of embryos), eggshell thinning, reduced clutch size, increased numbers of eggs laid outside of the nest, aberrant behavior of hatchlings, and potential hearing impairment in juveniles. In some cases, overall reproductive success in birds has decreased as much as 35 to 50 percent due to dietary methylmercury exposure insufficient to cause obvious signs of intoxication in adults. The most sensitive indicator of exposure appears to be reproductive parameters, which were used to establish the methylmercury TRVs for this BERA.

The avian TRVs were based on a three-generation study by Heinz (1974, 1976a,b, 1979) on mallard ducks. Mallards were fed methylmercury dicyandiamide at a level of 0.5 mg/kg-dw (0.1 mg/kg-ww) in dry duck mash. Females fed methylmercury laid fewer eggs and produced fewer ducklings than control ducks, and laid a greater number of eggs outside of nest boxes. Those ducklings that survived were less responsive to taped maternal warning calls and were hypersensitive to fright stimulus. Based on a food intake rate of 128 g/kg body weight (as reported by Heinz, 1979) for the treated F1 and F2 females, this represents a LOAEL TRV of 0.064 mg/kg body weight-day. No long-term studies were identified as suitable for the derivation of a no-effects TRV for methylmercury exposure to birds. Therefore, an uncertainty factor of 0.10 was applied to the LOAEL to derive a NOAEL of 0.0064 mg/kg-day.

Other avian field and laboratory studies support the concentration range of the TRVs selected. For example, Barr (1986) made similar observations in a field study of the common loon (*Gavia immer*) in northwestern Ontario. Egg laying and territorial fidelity were both reduced where mean mercury concentrations in loon prey was 0.3 to 0.4 mg/kg-ww. Loons in these areas established few territories and none laid any more than a single egg. The eggs contained mercury concentrations as high as 1.4 mg/kg-ww. Around waters where mean mercury concentrations of prey exceeded 0.4 mg/kg-ww, the loons raised no progeny. Reproductive effects may extend beyond the embryo and may reduce the rate of juvenile survival.

Subchronic histologic, neurologic, and immunologic effects were observed in great egrets dosed with methylmercury chloride at 0.5 mg/kg-ww, corresponding to intakes of 0.135 to 0.048 mg/kg-day during a 14-week experiment (Spalding et al., 2000). Dietary concentrations of methylmercury that produced significant reproductive impairment were about 20 percent of those required to produce overt neurological effects in adult birds (Scheuhammer, 1995).

The toxicity of methylmercury to mammals was based on two mink studies by Wobeser et al. (1976) and Wren et al. (1987). In a two-year study by Wobeser et al. (1976), mink were exposed to methylmercury chloride in their diets for 93 days (subchronic) at concentrations ranging from 1.1 to 15 ppm ww. Histopathological evidence of injury was present in all mink exposed to methylmercury. Clinical signs of neurotoxicity (anorexia and ataxia) were manifested at an exposure concentration of 1.8 ppm ww, and resulted in increased mortality in mink fed doses of 1.8 ppm or higher. In accordance with the procedures applied by USEPA in the Great Lakes Water Quality Initiative (USEPA, 1995b), an uncertainty factor of 0.1 was applied to account for extrapolation from a subchronic to a chronic toxicity. Based on a body mass of 1 kg for a mink in captivity and an intake rate of 0.137 kg ww/kg body weight per day (Bleavins and Aulerich, 1981), a NOAEL of 0.015 and a LOAEL TRV of 0.025 mg/kg bw-day were calculated from this study.

However, a study by Wren et al. (1987) observed increased mortality in mink fed 1 ppm of methylmercury for 81 days. The dosage was decreased after that time period due to excessive mortality. Wren et al. (1987) attributed the increased mortality to a combination of methylmercury exposure and cold stress, as the mink were maintained in outdoor cages. Based on these results, a NOAEL using the 1.1 ppm dosage from the Wobeser et al. (1976) study was not considered to be protective, as increased mortality occurs at lower dosages when combined with natural stresses present in field conditions. Therefore, an uncertainty factor of 0.1 was applied to the LOAEL of 0.025 mg/kg body weight-day to derive a NOAEL of 0.0025 mg/kg bw-day.

A study by Charbonneau et al. (1976), in which domestic cats (*F. Domesticus*) were exposed to methylmercury in their diets at doses ranging from 3 to 176 μg/kg-day continuously for up to two years, was not used to derive TRVs. In that study, significant and irreversible neurological impacts were noted at 74 μg/kg-day, while no significant neurological manifestations attributable to mercury exposure were observed at 46 μg/kg-day (0.046 mg/kg-day). Although cats and minks are both in the order Carnivora, they are in separate taxonomic suborders. *F. Domesticus* belong to suborder Feliformia, while mink and otter belong to suborder Caniformia. Both mink and otter are known to be very sensitive to the availability and toxicity of mercury within their habitat (Wren et al., 1986).

9.3.4.10 Nickel

Nickel is found in nature as a component of silicate, sulfide, or, occasionally, arsenide ores, and is usually found as Ni^{2+} in aquatic systems. Chemical factors that can affect the form of nickel in aquatic systems include pH and the presence of organic and inorganic ligands (USEPA, 1986d).

Nickel was selected as a COC for two avian receptor, the mallard and tree swallow, and one mammalian receptor, the little brown bat.

Reproductive and developmental effects from exposure to nickel have been observed in animals and various nickel compounds have been tested for mutagenicity (USEPA, 1986d). These tests have demonstrated the ability of nickel compounds to produce genotoxic effects; however, the translation of these effects into actual mutations is still not clearly understood. There is evidence both in humans and animals for the carcinogenicity of nickel, at least in some forms.

The avian TRVs were selected based on a study on mallard ducklings by Cain and Pafford (1981). Ducklings were fed nickel sulfate at three dose levels (176, 774, and 1,069 ppm) for 90 days. Consumption of up to 774 ppm nickel in the diet did not increase mortality or reduce growth; however, the 1,069 ppm nickel diet reduced growth and resulted in 70 percent mortality. Because the study considered exposure over 90 days, the 774 ppm dose was considered to be a chronic NOAEL and the 1,069 ppm dose was considered to be a chronic LOAEL. To estimate daily nickel intake throughout the 90-day study period, food consumption of 45-day-old ducklings was calculated. Using the consumption rate of 100 g food/day for a 1 kg adult (Heinz et al.,1989), a NOAEL of 77.4 mg/kg-day and a LOAEL of 107 mg/kg-day were calculated.

A study by Ambrose et al. (1976) was used to derive the mammalian TRVs. Rats were given doses of 250, 500, or 1,000 ppm nickel sulfate hexahydrate in their diets over three generations. While 1,000 ppm nickel in the diet reduced offspring body weights, no adverse effects were observed at the other dose levels. Because this study considered exposures over multiple generations, the 500 ppm dose was considered to be a chronic no effect level and the 1,000 ppm dose was considered to be a chronic lowest effect level. Assuming a body weight of 0.35 kg and a food ingestion rate of 28 g food/day (Sample et al., 1996), a NOAEL of 40 mg/kg-day and a LOAEL of 80 mg/kg-day were calculated.

9.3.4.11 Selenium

Selenium is a non-metallic element common in sedimentary soils. It is predominantly found either as insoluble metallic selenides or as soluble oxygen complexes, the most common being selenite and selenate. Average background concentrations in the US range from less than 0.1 to 4 ppm, with a mean of 0.31 (Kabata-Pendias and Pendias, 1992). Although selenium is an essential nutrient, exposure to high concentrations have been shown to result in adverse health effects. *In vivo*, selenium replaces sulfur in *de novo* amino acid synthesis, yielding selenomethionine and, to a lesser extent, selenocystine.

In birds, selenomethionine has been shown to be more toxic that the readily dissociated selenate or selenite. However, a survey of the mammalian literature yielded lower NOAELs for selenate than for any of the organic selenium compounds. The primary targets of toxicity include the gastrointestinal tract, the pancreas, and the thymus, with secondary toxicities associated with the kidney, liver, and central nervous system. Selenium has also been shown to be a reproductive toxicant, causing reduced fertility and increased malformations in both birds and mammals.

Selenium was selected as a COC for the tree swallow, belted kingfisher, great blue heron, and osprey, and for all mammalian receptors.

The toxicity of selenomethionine to birds was evaluated based on the results of a study by Heinz et al. (1989). Mallard ducks were fed 0, 1, 2, 4, 8, or 16 ppm selenomethionine in their diet for 100 days prior to egg set. An additional treatment of 16 mg/kg-day selenocystine was also included in the study. Reproductive productivity was significantly reduced at 8 ppm, with no significant effects noted at 4 ppm. Based on an average body weight of 1 kg and a food intake rate of 110 g dw/day (Heinz et al., 1989), a NOAEL TRV of 0.4 mg/kg-day and a LOAEL TRV of 0.8 mg/kg-day were derived.

The evaluation of the impact of selenium exposure on the reproduction of mammalian receptors was based on a study by Rosenfield and Beath (1954) in which rats were exposed to three levels of potassium selenate (1.5, 2.5, and 7.5 ppm) in drinking water over two generations. The treatment group exposed to 2.5 ppm showed no significant difference with regards to reduction rate or number of young reared. However, the second-generation female progeny of this treatment group showed a 50 percent reduction in the number of young reared. In the 7.5 mg/L group, fertility, juvenile growth, and survival were reduced. Therefore, the no-effects TRV was determined based on a dose of 1.5 ppm. Assuming a water intake rate of 0.046 L/day (based on the scaling function of Calder and Braun, 1983) and an average body weight of 0.35 kg (USEPA, 1988), a NOAEL TRV of 0.20 mg/kg-day and a LOAEL TRV of 0.33 mg/kg-day were determined.

9.3.4.12 Thallium

Thallium is a metal that can be released into the environment from coal combustion, heavy metals smelting, refining processes, and rodenticides. As a metal it exists in trace amounts in the earth's crust, with soil concentrations in the US ranging from 0.02 to 2.8 ppm (Kabata-Pendias and Pendias, 1992). The effects of thallium exposure can include gastroenteritis, diarrhea, constipation, vomiting, abdominal pain, and hair loss.

Thallium was selected as a COC for the tree swallow, little brown bat, and short-tailed shrew.

No avian studies on the effects of thallium were available and therefore no avian TRVs were derived for thallium.

The evaluation of thallium toxicity in mammals was based on a study by Formigli et al. (1986) in which rats were exposed to 10 ppm thallium sulfate (0.74 mg/kg-day, as provided in the study) in water for 60 days. The study dosage resulted in reduced sperm motility, and was therefore considered to be an LOAEL. Although exposure was subchronic, 90 percent of the animals in the treatment group showed effects of thallium exposure. Other endpoints, such as hair loss peripheral and nervous system disorders, have been documented to occur at the same dose (continuous exposure to 10 ppm thallium per day) in long-term studies (Manzo et al,.1983). Therefore, no uncertainty factors were applied for a LOAEL of 0.74 mg/kg-day. A NOAEL was estimated by applying a tenfold level of uncertainty to the LOAEL to derive a value of 0.074 mg/kg-day.

9.3.4.13 Vanadium

Vanadium is a natural constituent of soils, as well as being found in fuel oils and coal. Vanadium is concentrated mainly in mafic rocks and shales, and the average concentrations for US soils ranges between 58 and 100 ppm (Kabata-Pendias and Pendias, 1992). The most common anthropogenic sources involve vanadium entering the environment when fuel oils are burned.

Vanadium was selected as a COC for the tree swallow, mallard, little brown bat, short-tailed shrew, mink, and river otter.

The avian TRVs were developed from a study by White and Dieter (1978), in which mallard ducks were fed 2.84, 10.36, and 110 ppm vanadyl sulfate in their food over a 12-week period. No effects were observed at any dose level. The maximum dose of 11.4 mg/kg-day (based on study body weights and ingestion rates) was considered to be the chronic NOAEL. A chronic LOAEL of 114 mg/kg-day was calculated by applying an uncertainty factor of 10 to the NOAEL.

The mammalian TRVs were developed based on a study by Domingo et al. (1986), in which rats were exposed to sodium metavanadate ($NaVO_3$) at dose levels of 5, 10, and 20 mg/kg-day (corresponding to 2.1, 4.2, and 8.4 mg/kg bw-day) by oral intubation. Males were exposed for 60 days prior to mating and females were exposed for 14 days prior to mating. Significant decreases were observed in the development of the pups at all dose levels. The lowest dose (2.1 mg/kg bw-day $NaVO_3$) was therefore considered to be the chronic LOAEL TRV. The chronic TRV for mammals was determined by applying a tenfold level of uncertainty, to yield a NOAEL of 0.21 mg/kg-day.

9.3.4.14 Zinc

Zinc is used in many commercial products, including coatings to prevent rust, dry cell batteries, and is mixed with other metals to make alloys like brass and bronze. Some zinc is released into the environment by natural processes, but most comes from activities such as mining, steel production, coal burning, and burning of waste.

Zinc was selected as a COC for all avian receptors with an aquatic component in their diet (i.e., belted kingfisher, great blue heron, osprey, mallard, and tree swallow), the little brown bat, and the short-tailed shrew.

The avian TRVs were based on a study on leghorn hens by Stahl et al. (1990). The hens were fed zinc sulfate in their diet at doses of 48, 228, and 2,028 ppm for a period of 44 weeks. While no adverse effects were observed among hens consuming 48 and 228 ppm zinc, egg hatchability was less than 20 percent of controls among hens consuming 2,028 ppm zinc. The 228 ppm dose (corresponding to 131 mg/kg-day, based on the hens used in the study) was considered a chronic no-effect level and the 2,028 ppm dose was considered a chronic lowest effect level. Based on the body weights and food intake rates provided in the study, daily intake rates of 14.5 and 131 mg/kg-day were derived as the NOAEL and LOAEL.

The mammalian TRVs were based on a study by Schlicker and Cox (1968), where rats were exposed to zinc (2,000 and 4,000 mg/kg dose levels) in diet during gestation. Rats fed the higher dose displayed increased rates of fetal resorption and reduced fetal growth rates, while no effects were observed at the 2,000 mg/kg dose rate. As exposure occurred during gestation (a critical life stage), the lower dose corresponding to 160 mg/kg-day (based on body weight and intake rate) was considered a chronic NOAEL, and the higher dose corresponding to 160 mg/kg-day was considered to be a chronic LOAEL.

9.3.4.15 Bis(2-ethylhexyl)phthalate

Bis(2-ethylhexyl)phthalate (BEHP) is a synthetic chemical used principally as a plasticizer (an additive to plastics to make them more flexible), and may constitute as much as 40 percent of some PVC products (ATSDR, 1993). It is also used to a lesser extent in inks, pesticides, cosmetics, and vacuum pump oil (Sittig, 1991).

BEHP was selected as a COC for the tree swallow.

The avian TRVS were based on a study of ringed doves by Peakall (1974). Ringed doves were fed a dose of 10 ppm bis(2-ethylhexyl)phthalate (BEHP) in their diet over four weeks. No significant reproductive effects were observed among doves fed BEHP. The 10 ppm dose, corresponding to 1.1 mg/kg-d was considered to be a chronic NOAEL, as the study considered exposure over a critical life stage. A LOAEL of 11 mg/kg-day was derived by multiplying the NOAEL by an uncertainty factor of 10.

9.3.4.16 Chlordane

Chlordane is a viscous liquid, colorless to amber, with a slight chlorine-like aromatic odor. It was historically applied directly to soil or foliage (e.g., corn, citrus, deciduous fruits and nuts, and vegetables) to control a variety of insect pests. Short-term exposure to elevated levels of chlordane may affect the central nervous system, including irritability, excess salivation, labored breathing, tremors, convulsions, deep depression, and also result in blood system effects such as anemia and certain types of leukemia. Long-term exposure to chlordane has the potential to cause damage to liver, kidneys, heart, lungs, spleen and adrenal glands, and cancer.

Chlordane was selected as a COC for the short-tailed shrew.

The mammalian chlordane TRVs where based on Khasawinah and Grutsch (1989), in which mice (80/sex/group) were given 0, 1, 5, or 12.5 ppm technical chlordane in their diet for 104 weeks, corresponding to average doses of 0, 0.15, 0.75, and 1.875 mg/kg-day, respectively. Hematology, biochemistry, urinalysis, organ weights, and pathology of major tissues and organs were assessed on all animals that died during the study and on all survivors at week 104. Exposure-related effects were restricted to the liver. Based on the increased incidence of hepatic necrosis over controls, 1 ppm chlordane (0.15 mg/kg-day) was selected as the NOAEL, and 5 ppm chlordane (0.75 mg/kg-day) was selected as the LOAEL.

9.3.4.17 DDT and Metabolites

Avian species are particularly sensitive to the effects of DDT and its metabolites, specifically with regard to impacts on reproduction (McEwen and Stephenson, 1979). Toxicological impacts attributed to DDT exposure include eggshell thinning, reduced clutch size, elevated embryo mortalities, high mortality at time of pipping, increased hatchling mortality, and late nesting and unusual nesting behavior. In eggshell thinning, the activity of Ca^{2+} ATP-ase systems in the shell gland are affected, thereby interfering with the deposition of calcium in the shell (Lundholm, 1987). Eggshell thinning of greater than 20 percent has been associated with decreased nesting success due to eggshell breakage (Anderson and Hickey, 1969). Because of the tendency of DDT to magnify in food chains, higher trophic level birds appear to be at greater risk for egg loss due to shell thinning.

Another well-defined effect of DDT exposure is inhibition of acetylcholinesterase (AchE) activity. Inhibition of this enzyme results in the accumulation of acetylcholine in the nerve synapses, resulting in disrupted nerve function. Chronic inhibition of 50 percent of brain AchE has been associated with mortality in birds (Ludke et al., 1975).

The effects of DDT on other receptor groups are not as clearly defined. Recent studies indicate that DDT may be an estrogenic mimic, resulting in adverse reproductive effects. Observed effects include feminization and increased female:male population ratios for some receptors. Other responses include histopathological changes, alterations in thyroid function and changes in the activity of various enzyme groups (Peakall, 1993). In addition to toxic effects, DDT and its metabolites can bioaccumulate.

DDT and its metabolites was selected as a COC for the tree swallow, belted kingfisher, great blue heron, osprey, red-tailed hawk, and mink.

The TRV used for the evaluation of toxic effects of DDT and its metabolites in birds was based on the results of Anderson et al. (1975), which is the same study used by the Great Lakes Water Quality Initiative (USEPA, 1995b). Anderson et al. (1975) studied the reproductive success of brown pelicans (*Pelecanus occidentalis*) off the coast of southern California from 1969 through 1974. Concentrations of DDT and its metabolites in northern anchovies (a main component of the brown pelicans' diet) and pelican eggs were monitored during the course of the five-year investigation. Over this time, total DDT (and metabolite) concentrations declined in the fish from 4.27 to 0.15 ppm ww (0.60 ppm dw). At the lowest prey concentration, the fledgling rate was still 30 percent below that needed to maintain a stable population (1.2 to 1.5 young per pair). Therefore, 0.15 ppm was considered as the LOAEL. An LOAEL TRV of 0.028 mg/kg body weight-day was derived for birds, based on a body mass of 3.5 kg for an adult brown pelican (Dunning, 1993), and a food ingestion rate of 0.66 kg/day (USEPA, 1995b). A NOAEL TRV of 0.0028 mg/kg-day was derived by applying an uncertainty factor of 0.1.

For mammals, the lowest DDT and metabolite TRVs were derived from a reproductive study by Fitzhugh (1948), in which rats were exposed to doses of 10, 50, 100, and 600 ppm in their food for two years. While consumption of 50 ppm or more DDT in the diet reduced the number of young produced, no adverse effects were observed at the 10 ppm DDT dose level. These doses correspond to LOAELs and NOAELs of 0.8 and 4.0 mg/kg-day, respectively. It should be noted that studies indicate that mustelids (the family that mink belong to) can rapidly degrade DDT (e.g., Roos et al., 2001); therefore, the mammalian TRVs may be conservative for mink.

9.3.4.18 Dichlorobenzenes

1,2-Dichlorobenzene is used mainly as a chemical intermediate for making agricultural chemicals, primarily herbicides. Other present and past uses include use as a solvent for waxes, gums, resins, wood preservatives, and paints; insecticide for termites and borers; in making dyes; as a coolant, deodorizer, and de-greaser. 1,4-Dichlorobenzene (p-DCB) is an organic solid of white crystals with a mothball-like odor. It is used mainly as an insecticidal fumigant against moths in clothes and as a deodorant for garbage and restrooms. It is also used as an insecticide and fungicide on crops, in the manufacture of other organic chemicals, and in plastics, dyes, pharmaceuticals. Dichlorobenzene is known to bioaccumulate because of rapid metabolic turnover in exposed organisms. The long-term exposure to dichlorobenzenes may result in damage to the liver, kidneys, and cellular components of the blood, and may cause anemia, skin lesions, and appetite loss. Some neurological effects have been linked to inhalation of dichlorobenzene.

Dichlorobenzenes were selected as a COC for the mallard and the tree swallow.

The toxicity of dichlorobenzene to birds was evaluated based on a feeding study by Hollingsworth et al. (1956) in which geese were exposed to 500 ppm *p*-dichlorobenzene in their diet for a duration of five weeks. An exposure dose based on the measured intake rate was estimated to be approximately 600 mg/kg-day. The toxicological impacts of this exposure included a general reduction in growth and a mortality rate of 30 percent. This was therefore considered to be a sub-chronic LOAEL TRV. A tenfold uncertainty factor was applied to convert this value into a chronic LOAEL of 60 mg/kg-day, and an additional uncertainty factor of 0.1 was applied to derive a chronic NOAEL value of 6 mg/kg-day.

9.3.4.19 Dieldrin

Dieldrin, a chlorinated insecticide, was widely used from the 1950s to the 1970s, for soil and seed treatment, to control mosquitos and tsetse flies, as a sheep dip, for wood treatment, and for mothproofing woolen products. Most uses of dieldrin were banned in 1975, and it is no longer produced in, or imported to, the US (ASTDR, 1998). Dieldrin's toxic effects include carcinogenicity, mutagenicity, neurotoxicity, teratogenicity, and reproductive impairment.

Dieldrin was selected as a COC for all mammalian receptors.

In mammals, dieldrin is rapidly absorbed from the gastrointestinal tract upon ingestion. It is then transported from the liver to various tissues in the body, including the brain, blood, liver, and adipose tissue. Toxicity appears to be related to the central nervous system, resulting in stimulation, hyperexcitability, hyperactivity, incoordination, and exaggerated body movement, ultimately leading to confusion, depression, and death (ASTDR, 1998). Dieldrin has been shown to cross the placental barrier, and for that reason has been studied for its teratogenic properties and reproductive effects.

In the study selected for deriving TRVs, rats were exposed to dietary concentrations of dieldrin ranging from 0.08 to 40 mg/kg for up to 336 days (Harr et al., 1970). The concentration of 0.31 ppm (0.018 mg/kg-day) was the lowest concentration that resulted in adverse reproductive effects, which included a reduction in pup survival and conception rate. The highest dose that did not produce any reproductive effects was 0.16 ppm (0.009 mg/kg-day). Therefore, a value of 0.018 mg/kg-day was selected as the LOAEL and 0.009 mg/kg-day was selected as the NOAEL.

9.3.4.20 Dioxins/Furans

Polychlorinated dibenzo-*p*-dioxins (PCDDs) are composed of a triple-ring structure consisting of two benzene rings connected to each other by two oxygen atoms. Depending on the number and position of chlorine substitution on the benzene rings, 75 chlorinated dioxin congeners are possible. The polychlorinated dibenzofuran (PCDF) molecule is also a triple-ring structure, with the two benzene rings connected to themselves by a single oxygen atom. In all, 135 chlorinated dibenzofuran congeners are possible.

Dioxins and furans are not produced intentionally, but are unavoidable byproducts of chemical manufacturing or the result of incomplete combustion of materials containing chlorine atoms and organic compounds. Dioxins and furans may also be formed during the disinfection of complex effluents (e.g., pulp and paper effluents) containing many organic constituents.

Dioxins and furans may be distributed throughout the environment via air, water, soil, and sediments. Dioxins and furans tend to be very insoluble in water, adsorb strongly onto soils, sediments, and airborne particulates, and bioaccumulate in biological tissues (Hutzinger et al., 1985). These substances have been associated with a wide variety of toxic effects in animals, including acute toxicity, enzyme activation, tissue damage, developmental abnormalities, and cancer.

Dioxins, like PCBs, are polychlorinated hydrocarbons (PCHs) and toxicity is believed to be mediated intracellularly by binding with the aryl hydrocarbon receptor (AhR). The resulting PCH-AhR complex moves into the cell nucleus, where it will bind to the DNA, and may alter the expression of a number of gene sequences. Many of the observed toxic effects of dioxins (and the coplanar PCBs) are attributable to specific alterations in gene expression.

The effects of tetrachlorodibenzo-*p*-dioxins (TCDDs) have been reviewed by Safe (1990) and Giesy et al. (1994). Dioxins are not generally acutely toxic to adult organisms, but their long-term accumulation is thought to be expressed chronically, and may ultimately result in death. Key effects are those causing reproductive dysfunction. The PCDDs and PCDFs are thought to cause alterations to developmental endocrine functions (thyroid and steroid hormones), as well as interference in vitamin production, which results in disruption of patterns of embryonic development at critical stages (Giesy et al., 1994). General population-level manifestations of dioxin exposure include adversely affected patterns of survival, reproduction, growth, and resistance to diseases (Eisler and Belisle, 1996). Poor reproductive efficiencies and adventive, opportunistic diseases are characteristic of wild animals in the exposed populations of the Great Lakes region (Giesy et al., 1994).

To assess toxicity, chlorinated dioxins and furans are classified at varying levels of potency of 2,3,7,8-TCDD (Eastern Research Group [ERG], 1998). These variations in potency are quantified based on receptor-specific TCDD toxicity equivalence factors (TEFs).

Dioxins/furans were selected as a COC for all receptors, except for the great blue heron.

A study by Nosek et al. (1992) was used to derive avian TRVs. Ring-neck pheasants (*Phasianus colchicus*) were exposed to 2,3,7,8-TCDD at three dose levels: 0.01, 0.1, and 1.0 μp/kg BW/week via weekly intraperitoneal injection (equivalent to 1.4×10^{-6}, 1.4×10^{-5}, and 1.4×10^{-4} mg/kg-day) for 10 weeks, through reproduction. No adverse effects on reproduction were observed at the two lower dose levels. Therefore, the highest dose level of 1.4×10^{-4} mg/kg-day was considered to be a chronic LOAEL TRV and the 1.4×10^{-5} mg/kg-day the NOAEL TRV for birds.

The mammalian TRV used for the evaluation of risk to mammals was based on a study by Murray et al. (1979), who exposed rats to TCDD at three dose levels (1×10^{-6}, 1×10^{-5}, and 1×10^{-4} mg/kg-day in food) for three generations. The 1×10^{-6} mg/kg-day dose resulted in no significant adverse effects, and was therefore considered to be a chronic NOAEL TRV for mammals. The 1×10^{-5} mg/kg-day dose resulted in an approximately 10 percent reduction in live births and was therefore applied as the LOAEL TRV.

9.3.4.21 Endrin

Endrin is an organochlorine insecticide which has been used since the 1950s against a wide range of agricultural pests, mostly on cotton but also on rice, sugar-cane, maize, and other crops. It is also used as a rodenticide.. Like other chlorinated hydrocarbon insecticides, endrin also affects the liver, and stimulation of enzyme systems involved in the metabolism of other chemicals (WHO, 1992).

Endrin was selected as a COC for the belted kingfisher.

A study by Fleming et al. (1982) on screech owls (*Otus asio*) fed 0.75 ppm endrin in their diet for 83 days was used to derive the avian TRVs. Egg production and hatching success were reduced among owls fed endrin. Because the study considered exposure throughout a critical life stage (reproduction), the normalized dose of 0.1 mg/kg-d was considered to be a chronic LOAEL. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL to NOAEL uncertainty factor of 0.1 to obtain a value of 0.01 mg/kg-d.

9.3.4.22 Hexachlorobenzene

Hexachlorobenzene was widely used as a pesticide and fungicide for onions and wheat and other grains until 1965. It was also used in the manufacture of fireworks, ammunition, electrodes, dye, and synthetic rubber, and as a wood preservative (Sitting, 1991; ATSDR, 1997). There are currently no commercial uses of hexachlorobenzene (ATSDR, 1997).

Hexachlorobenzene was selected as a COC for the mink, little brown bat, and short-tailed shrew.

The mammalian TRVs were based on a study of mink and European ferrets (*Mustela putorius furo*) by Bleavins et al. (1984) fed diets that contained hexachlorobenzene (HCB). Diets treated with 125 or 625 ppm HCB were lethal to the adults of both species. The cross-fostering of mink kits whelped by untreated dams to females fed 2.5 mg/kg HCB resulted in increased kit mortality when compared to untreated controls. The in utero exposure to HCB resulted in higher kit mortality than exposure via the dam's milk. This dose resulted in a LOAEL of 0.14 mg/kg-day and a factor or 0.1 was applied to yield a NOAEL of 0.014 mg/kg-day.

9.3.4.23 Hexachlorocyclohexanes

Hexachlorocyclohexanes are found in the organochlorine insecticide lindane. The solubility of hexachlorocyclohexane isomers in lipid are as follows: delta > gamma > alpha > beta. Lindane has not been produced in the US since 1977, although it is still imported into and formulated in the US. Former uses included insecticide on fruit and vegetable crops including greenhouse vegetables and forest crops including Christmas trees.

Hexachlorocyclohexanes were selected as a COC for the belted kingfisher, great blue heron, and osprey.

Jansen's (1996) study on the common quail (*Cotumix cotumix*) was used to derive avian TRVs. Lindane was dissolved in the drinking water of captive quail at doses of 1, 3, and 9 ppm, for seven days. Eggshell thickness, egg volume, egg mass, incubation time, hatchability, and embryo development were recorded prior to, during, and after the treatment. Egg production was not affected by exposure to lindane. Egg mass was reduced significantly and egg volume increased slightly at 3 ppm lindane. There was no significant eggshell thinning as a result of exposure to lindane. Fertility and hatchability were lower at 3 and 9 ppm of lindane and incubation period was slightly reduced and overall fecundity decreased as a result of lindane ingestion. Doses of 1 and 3 ppm were selected as the NOAEL and LOAEL, respectively. Although quail were only exposed for seven days, exposure occurred during a sensitive reproductive period and therefore no uncertainty factor was applied. Using a body weight of 0.15 kg, based on Vos et al. (1971), a food intake rate of 16.9 g dw/day (Nagy, 1987), a NOAEL of 0.11 mg/kg-day and a LOAEL of 0.34 mg/kg-day were derived.

9.3.4.24 Polychlorinated Biphenyls

PCBs are industrial compounds that were used in a broad range of commercial applications until their manufacture was banned in 1976 under the Toxic Substances Control Act (TSCA) (15 U.S.C. Sec. 2601 et seq.). They are complex chemicals consisting of ten different homolog classes (monochlorobiphenyls to decachlorobiphenyls) that are distinguished by the number of chlorine atoms bound to the biphenyl molecule. Among these ten homologs are 209 different PCB congeners, which reflect the different number and location (isomer) of the bound chlorine atoms. PCBs were manufactured in the US under several trade names, but they are best identified with the name Aroclor (manufactured by Monsanto). Different Aroclors, which consist of different mixtures of congeners, were given four-digit codes (e.g., Aroclors 1248, 1260); the last two digits usually indicate the chlorine content (by percent weight).

For reviews of PCB toxicology, see Seegal (1996), Tilson et al. (1990), Safe (1990, 1994), Kimbrough (1985), Silberhorn et al. (1990), Bolger (1993), and Delzell et al. (1994).

PCBs were selected as a COC for all receptors, with the exception of the red-tailed hawk.

The avian TRVs selected for this BERA are based on a 16-week study by Dahlgren et al. (1972) that examined the effects of Aroclor 1254 on pheasants. In this study, ring-neck pheasants were dosed once a week with either 12.5 or 50 mg/bird-week Aroclor 1254. No impact on chick growth, egg production, or survivability was reported at the lower dose; however, egg hatchability was slightly lower at this dose. Therefore, the 12.5 mg/bird-week dose, equal to a daily dose of 1.8 mg/kg-day, was considered a LOAEL TRV. A NOAEL was derived by multiplying the LOAEL by a factor of 0.1 for a value of 0.18 mg/kg-day.

For mammalian receptors, separate TRVs were derived for mustelid receptors (mink and otter) and other mammals (bat and shrew), as mustelids are known to be sensitive to PCBs. A multigenerational study on mink by Restum et al. (1998) was used to derive mustelid TRVs. Over a three-year period, captive mink were fed carp caught from Saginaw Bay (Lake Superior) that contained PCBs and other contaminants. These fish were mixed with clean fish to produce treatment diets of 0, 0.25, 0.5, and 1 ppm ww total PCBs. Continuous exposure to 0.25 ppm or more delayed the onset of estrus and lessened the whelping (birth) rate. It also resulted in significantly lower body weights of kits at six weeks. Based on these effects, the 0.25 ppm treatment was considered a LOAEL. The LOAEL TRV was calculated to be 0.034 mg/kg-day, based on a body mass of 1 kg and an intake rate of 0.137 kg ww/kg body weight per day (Bleavins and Aulerich, 1981). The NOAEL TRV was estimated by the application of an uncertainty factor of 0.1 for a value of 0.0034.

The TRVs for the little brown bat and short-tailed shrew were based on multigenerational study of rats by Linder et al. (1974). Mating pairs of rats and their offspring were fed Aroclor 1254 at concentrations of 5 and 20 ppm. Offspring of rats fed 20 ppm exhibited decreased litter size (reduction of 15 to 24 percent) in comparison to controls., while those fed 5 ppm did not significantly differ from controls. Using a body weight of 0.35 kg and a food intake rate of 28 g/day based on Sample et al. (1996), a NOAEL of 0.4 mg/kg-day and a LOAEL of 1.6 mg/kg-day were calculated.

9.3.4.25 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) is the general term applied to a group of compounds comprised of several hundred organic substances with two or more benzene rings. They occur in the environment mainly as a result of incomplete combustion of organic matter (forest fires, internal combustion engines, wood stoves, coal, coke, etc.). They are major constituents of petroleum and its derivatives. In addition, wastewater treatment plant effluents and runoff from urban areas, particularly from roads, are known to contain significant quantities of PAHs. Inputs of PAHs in aquatic ecosystems may occur as a result of oil spills, forest fires and agricultural burning, leaching from waste disposal sites, and coal gasification (Eisler, 1987b; Neff, 1979). PAHs are also produced by natural processes at very low rates.

In aquatic environments, PAHs tend to form associations with suspended and deposited particulate matter (Eisler, 1987b). This sorption of PAHs to sediments is strongly correlated with the TOC content of sediments, which influences its bioavailability. In general, elevated levels of sediment-associated PAHs are found in the vicinity of urban areas. Exposure to PAHs may result in a wide range of effects on biological organisms. While some PAHs are known to be carcinogenic, others display little or no carcinogenic, mutagenic, or teratogenic activity (Neff, 1979). Many carcinogenic PAHs also exhibit teratogenic and mutagenic effects. Several PAHs exhibit low levels of toxicity to terrestrial life forms, yet are highly toxic to aquatic organisms (Eisler, 1987b). Although PAHs are taken up and accumulated in terrestrial and aquatic plants, fish, and invertebrates, bioconcentration is limited by metabolism and elimination in many species.

PAHs can interact with cells in two ways to cause toxic responses. They may bind reversibly to lipophilic sites in the cell and thereby interfere with several cellular processes. Alternatively, their metabolites may bind covalently to cellular structures, causing long-term damage.

PAHs were selected as COC for all wildlife receptors except the osprey. Studies on the toxicity of PAHs in birds, particularly with regard to impacts on reproduction, are rare. A study examining the embryotoxicity of an artificial mixture of 18 PAHs in chicken, turkey, domestic duck, and common eider (*Somateria mollissima*) eggs found that a dose of 2 mg/kg-egg increased mortality among the embryos of all four species, and mortality was also increased in the duck at 0.2 mg/kg-egg (Brunstrom et al., 1990). Hough et al. (1993) examined the effects of benzo[a]pyrene on pigeons. Three- to six-month-old pigeons were administered a dose of 10 mg/kg weekly for a period of five months. The treatment birds were reported to have suffered complete reproductive failure and an associated gross alteration in ovarian structure. This dose, which corresponds to a daily exposure of 1.43 mg/kg-day, was considered representative of an LOAEL for birds. To estimate the NOAEL TRV, a tenfold level of uncertainty was applied to the LOAEL TRV to derive an estimate of 0.143 mg/kg-day.

The evaluation of PAH toxicity to mammals was based on a study by Mackenzie and Angevine (1981) that examined the reproductive effects of benzo[a]pyrene on mice. Female mice were exposed to benzo[a]pyrene at doses of 10, 40, and 160 mg/kg-day through daily intubation. Treatment commenced on day 7 after the best estimated time of conception and continued through day 16 of gestation. The effects of exposure were followed for up to six months. Although the duration of exposure was short, it is considered to be a chronic study because exposure occurred during a critical life stage. Total sterility was observed in 97 percent of the mice exposed prenatally to 40 or 160 mg/kg benzo[a]pyrene. Fertility was markedly impaired in animals exposed *in utero* to 10 mg/kg benzo[a]pyrene. After six months on a breeding study female mice in this group gave birth to significantly fewer and smaller litters and male mice in this group impregnated 35 percent fewer females than controls. The 10 mg/kg-day treatment was therefore considered to be applicable as a LOAEL TRV. The estimation of the NOAEL TRV was based on the application of a tenfold level of uncertainty to the toxicity estimate to derive a value of 1 mg/kg-day.

9.3.4.26 Trichlorobenzenes

1,2,4-Trichlorobenzene is an aromatic, colorless, organic liquid that is used primarily as a dye carrier. It is also used to make herbicides and other organic chemicals, as a solvent, in wood preservatives, and, previously, as a soil treatment for termite control. Short-term effects of exposure to 1,2,4-trichlorobenzene may cause changes in the liver, kidneys, and adrenal glands, and long-term exposure may result in increased adrenal gland weights.

Trichlorobenzenes were selected as a COC for the tree swallow, mallard, little brown bat, and short-tailed shrew.

No avian studies were found on the toxicity of trichlorobenzenes; therefore, potential risks to birds from this COC will be addressed in the BERA only qualitatively.

The mammalian TRVs are based on a multigenerational rat study (Robinson et al., 1981). From birth, male and female rats (F0 generation) were dosed with 0, 25, 100, or 400 ppm of 1,2,4-trichlorobenzene in the drinking water. Similar procedures were performed with their offspring (F1 generation) and the F2 generation. Fertility, as measured by the conception rates of the females, of F0 and F1 generation rats was not affected by treatment. An LOAEL was derived from a significant increase (11 percent in males, 13 percent in females) in adrenal gland weights observed in the 400 ppm (53.6 mg/kg-day) groups of males and females of the F0 and F1 generations. The NOAEL was considered to be the 100 ppm (14.8 mg/kg-day) dose. Effects on the F2 generation were less than on the F0 and F1 generations.

9.3.4.27 Xylenes

Xylenes are clear liquids with a sweet odor that are used mainly as a solvent. Other uses include as a component of gasoline, and for the production of phthalate plasticizers, polyester fiber, film, and fabricated items. Short-term exposure may result in disturbances of cognitive abilities, balance, and coordination, while long-term exposure may cause damage to the central nervous system, liver, and kidneys.

Xylenes were selected as a COC for the mallard, tree swallow, and little brown bat.

No studies examining the toxicity of xylenes to birds were found; therefore, potential risks to birds from this COC will be addressed in the BERA only qualitatively.

The mammalian TRVs are based on a mouse study by Marks et al. (1982), where mice received doses of 0.5, 1.0, 2.1, 2.6, 3.1, or 4.1 mg/kg-d of mixed xylenes a xylene mixture (60 percent m-xylene, 9 percent o-xylene, 14 percent p-xylene and 17 percent ethylbenzene) from days 6 to 15 of their gestation period via oral gavage. Xylene exposure of 2.6 mg/kg/d or greater significantly reduced fetal weights and increased the incidence of fetal malformities. While the xylene exposures evaluated in this study were of a short duration, they occurred during a critical life stage. Therefore, the highest dose that produced no adverse effects, 2.1 mg/kg/d, was considered to be a chronic NOAEL and the 2.6 mg/kg/d dose level was considered to be a chronic LOAEL.

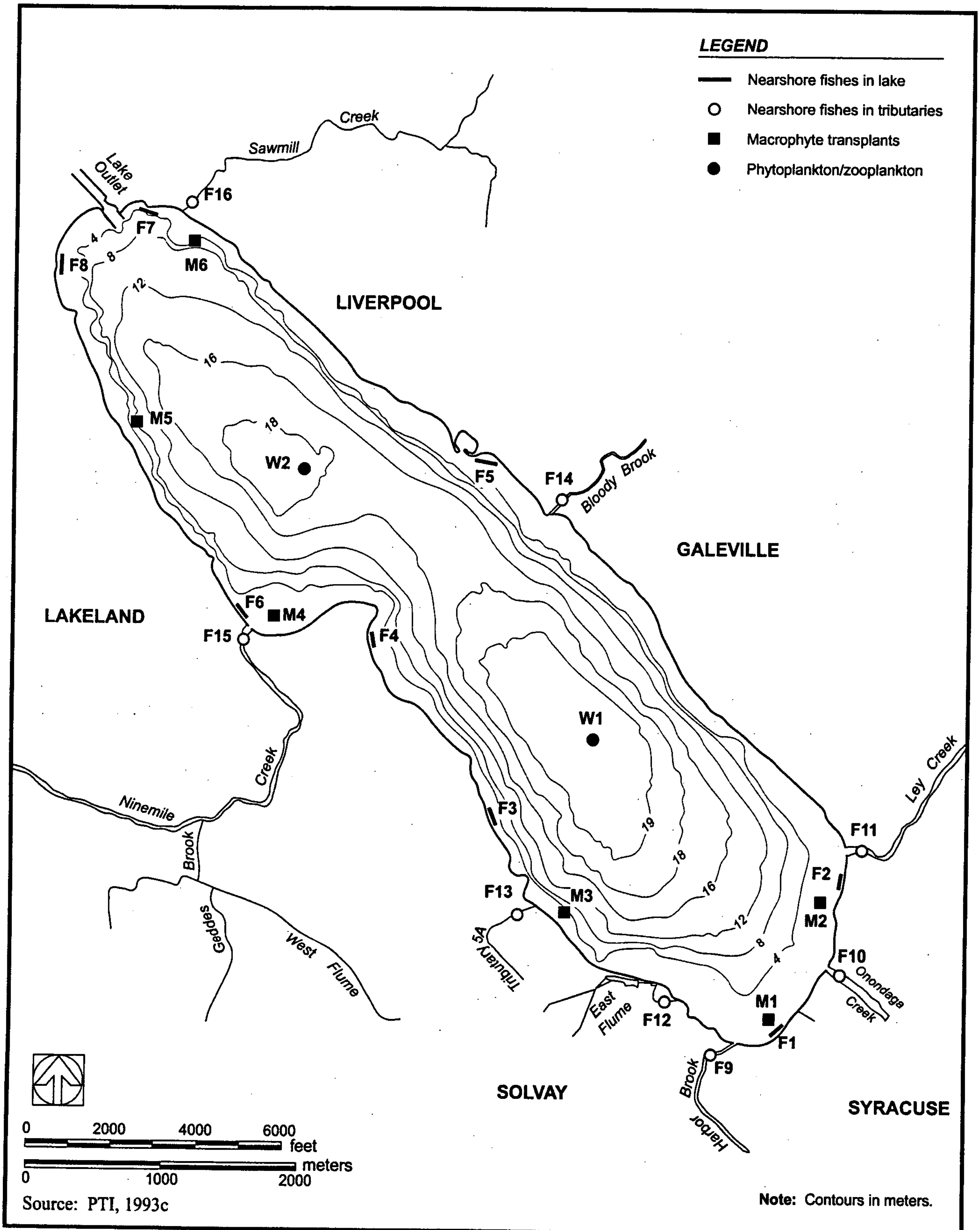
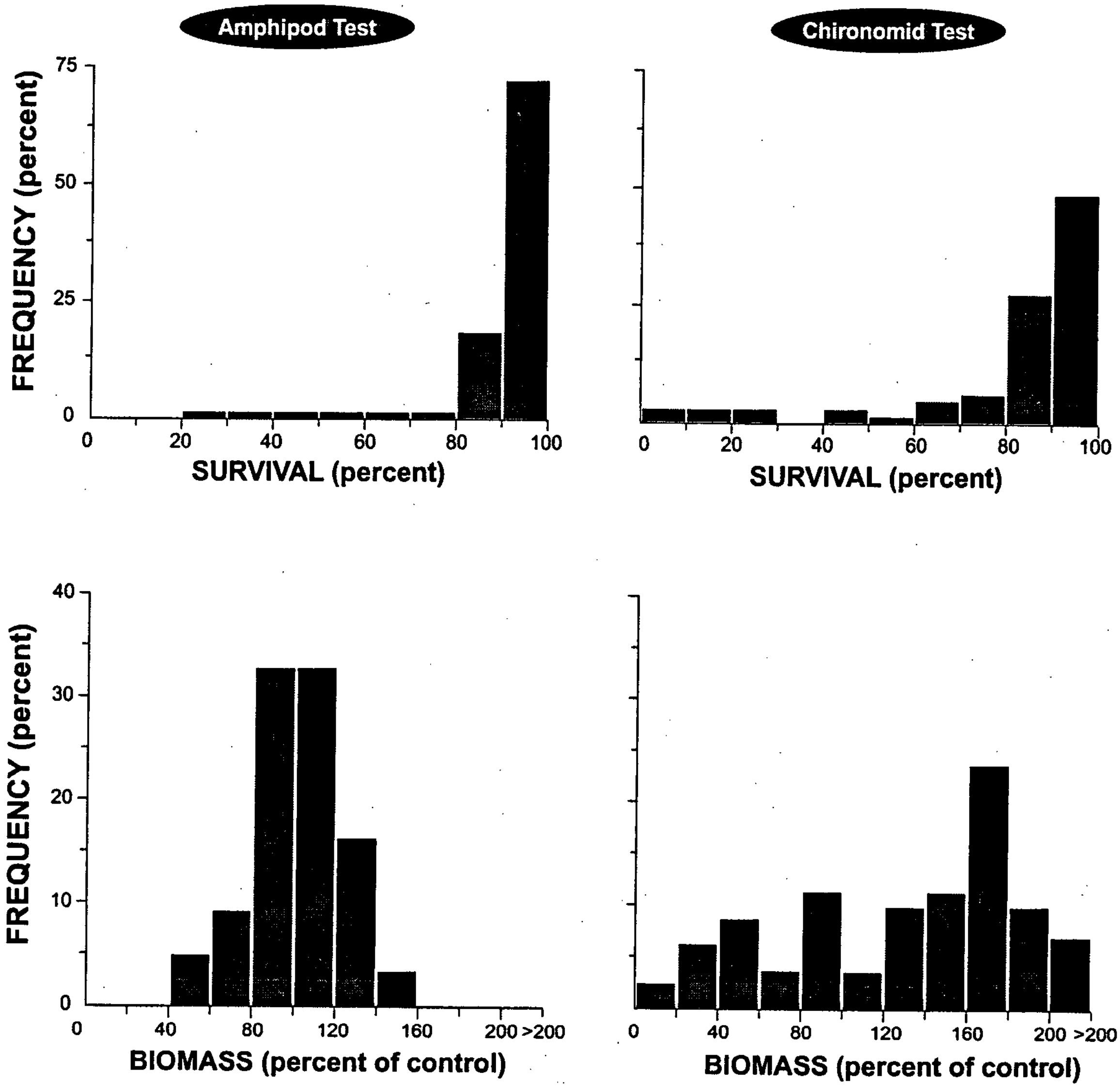


Figure 9-1. Locations of stations for the nearshore fish, macrophyte transplant, and phytoplankton/zooplankton studies of Onondaga Lake and its tributaries in 1992



Source: Exponent, 2001b

Figure 9-2. Frequency distributions of survival and biomass for amphipod and chironomid 10-day sediment toxicity tests for Onondaga Lake in 1992

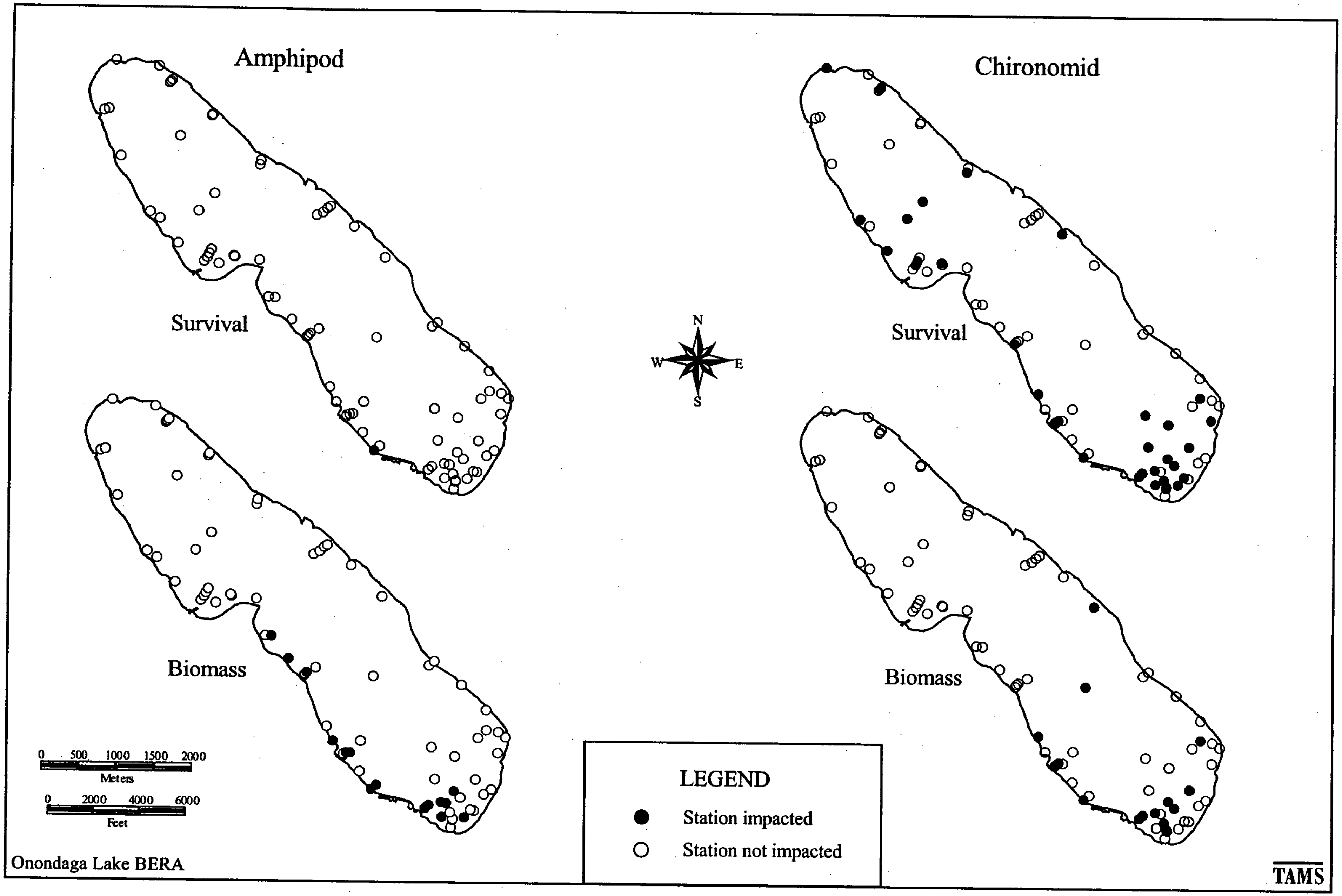


Figure 9-3
Locations of Stations in Onondaga Lake at Which Significant Toxicity Was Found Using the 10-day Amphipod and Chioronomid Tests in 1992

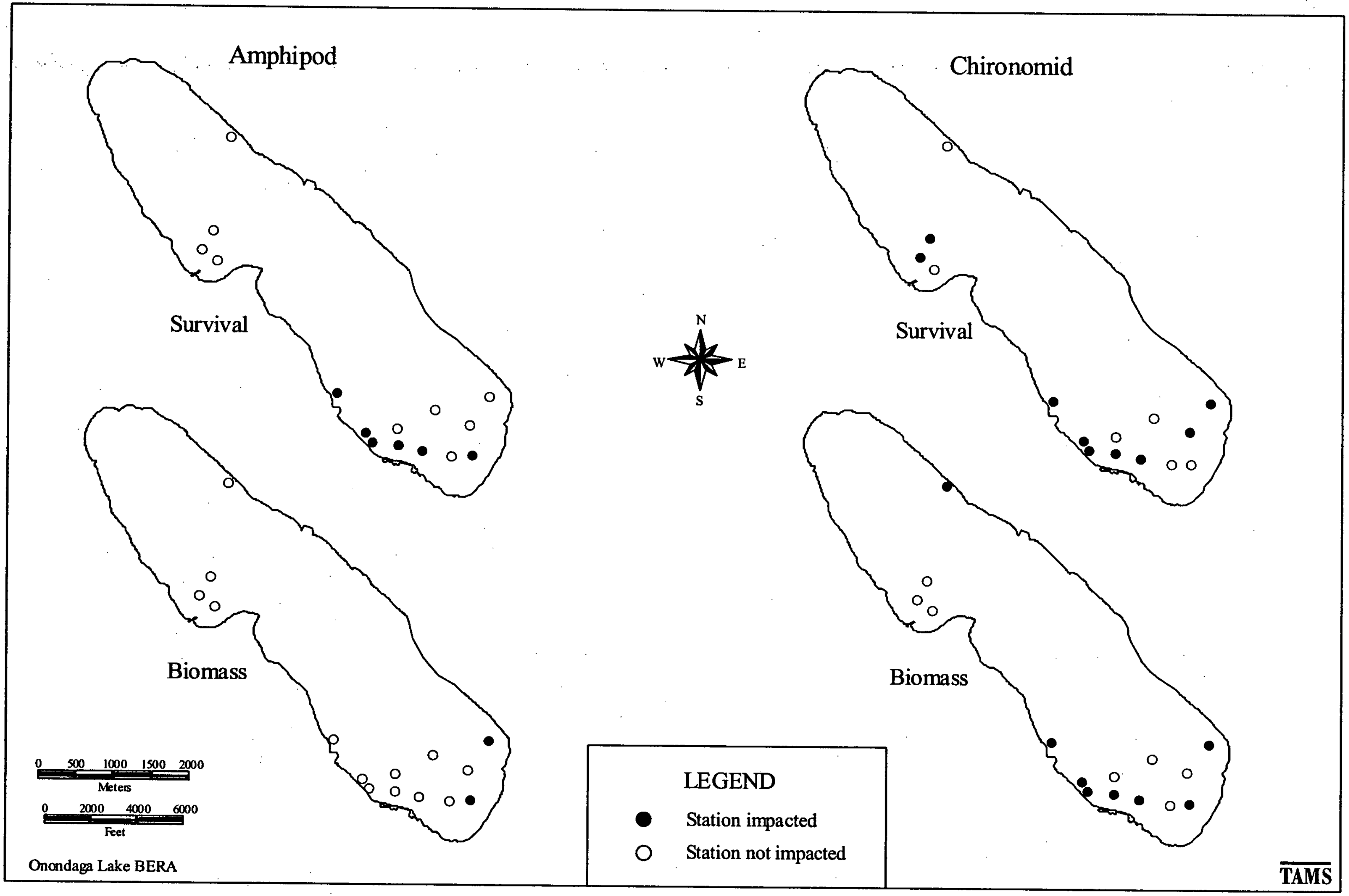
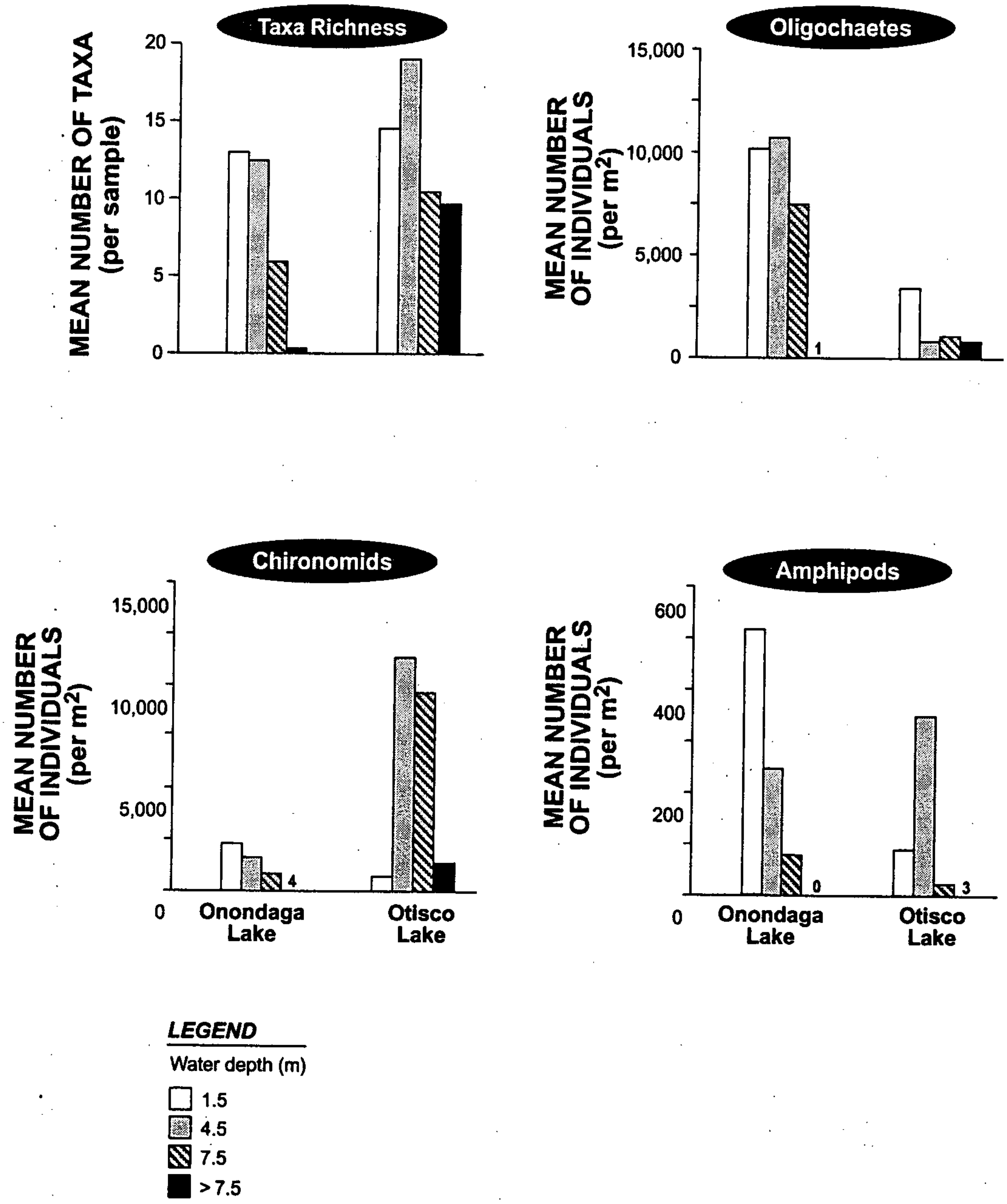
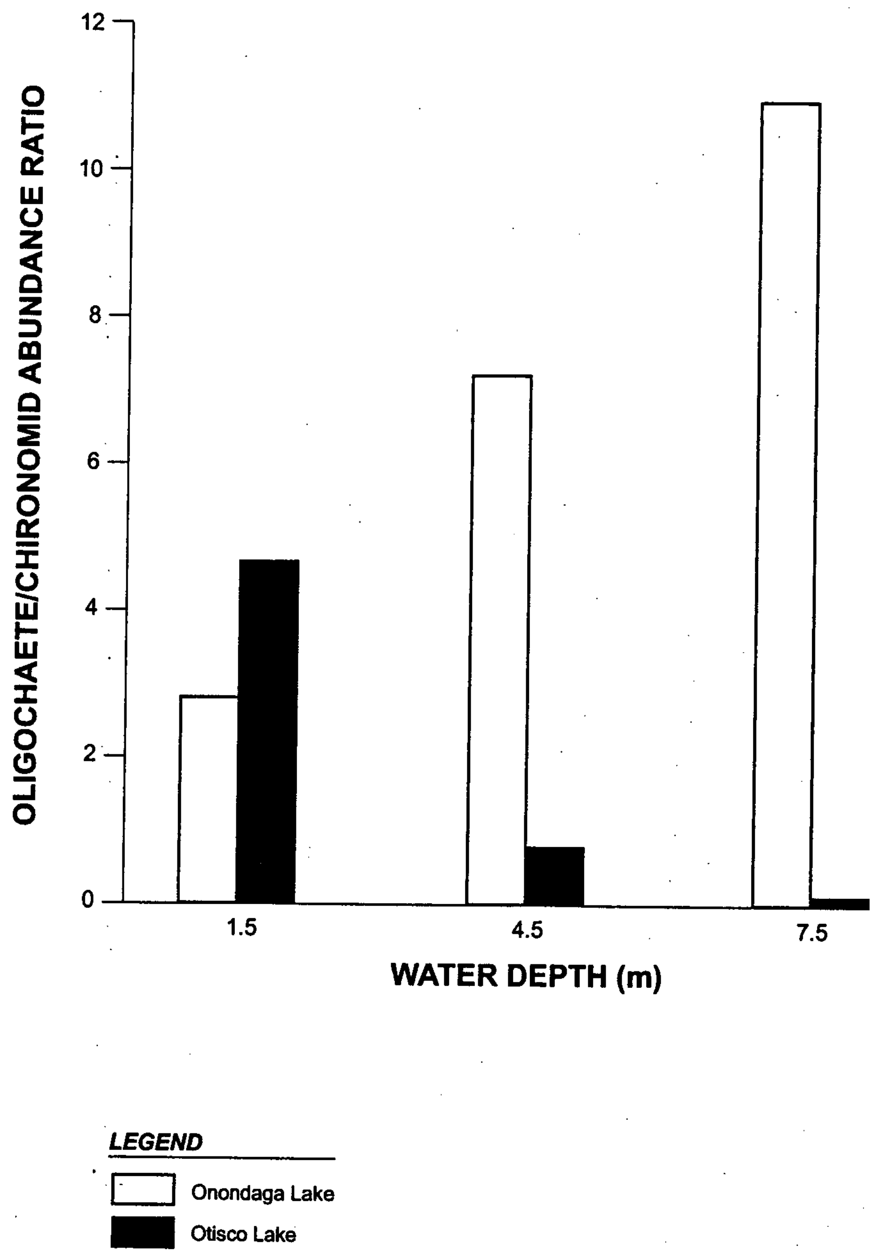


Figure 9-4
Locations of Stations in Onondaga Lake at Which Significant Toxicity Was Found Using the 42-day Amphipod and Chironomid Tests in 2000



Source: Exponent, 2001b

Figure 9-5. Comparison of major benthic macroinvertebrate community variables among Onondaga and Otisco lakes in 1992



Source: Exponent, 2001b

Figure 9-6. Comparison of oligochaete/chironomid abundance ratios among Onondaga and Otisco lakes in 1992

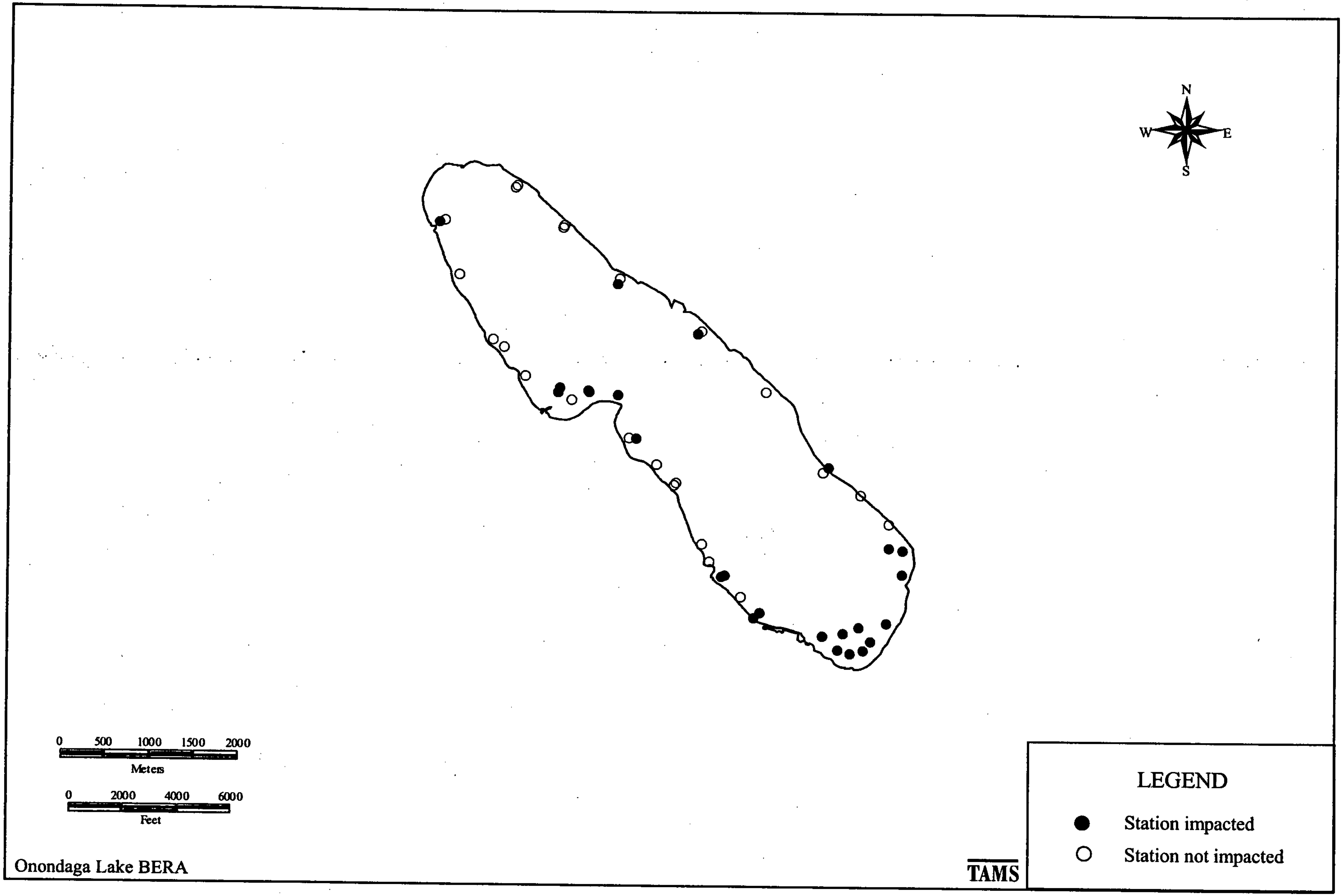


Figure 9-7
Patterns of Benthic Taxa Richness in Onondaga Lake in 1992

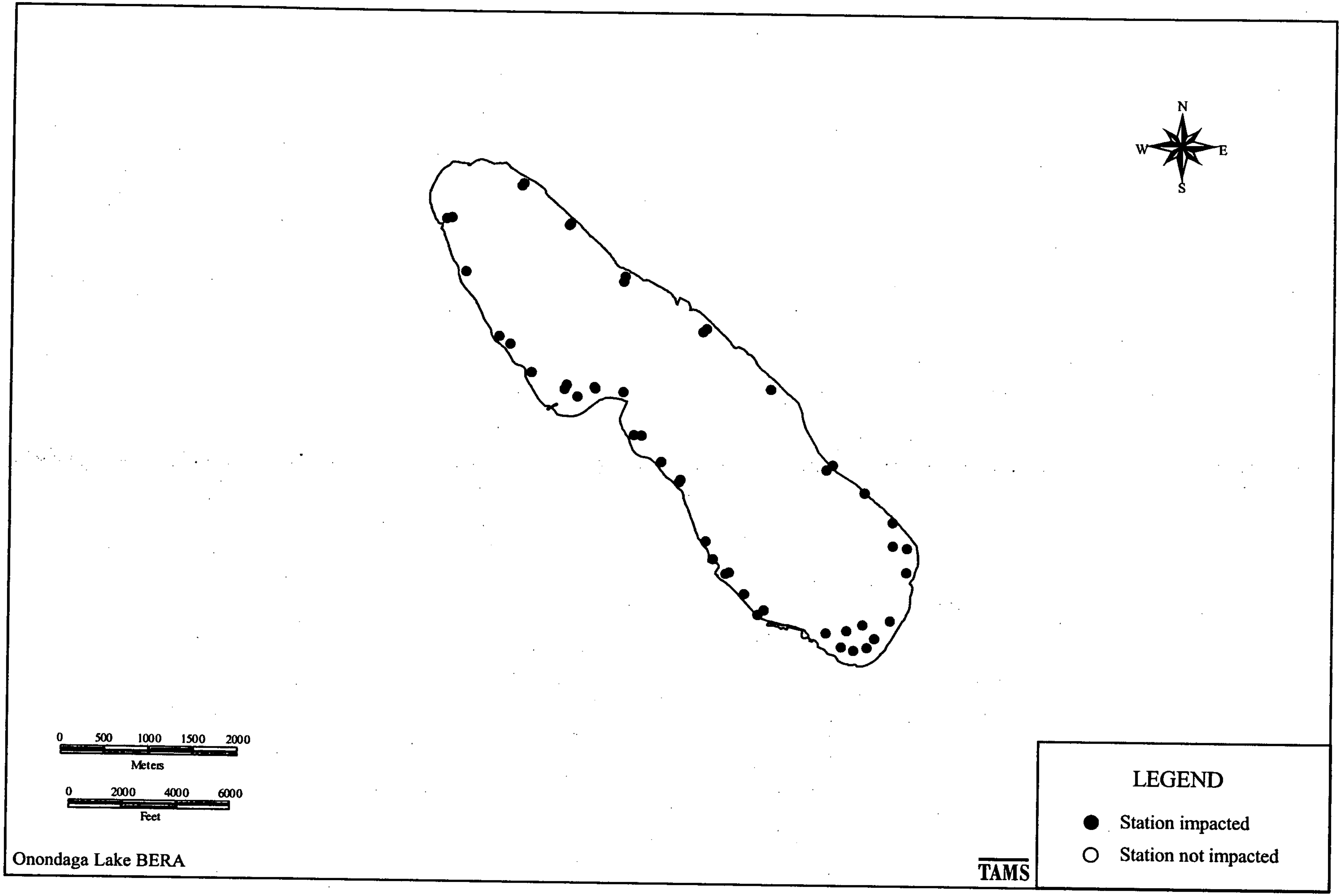


Figure 9-8
Patterns of Richness of Non-Chironomidae/Oligochaeta (NCO) Taxa Richness in Onondaga Lake in 1992

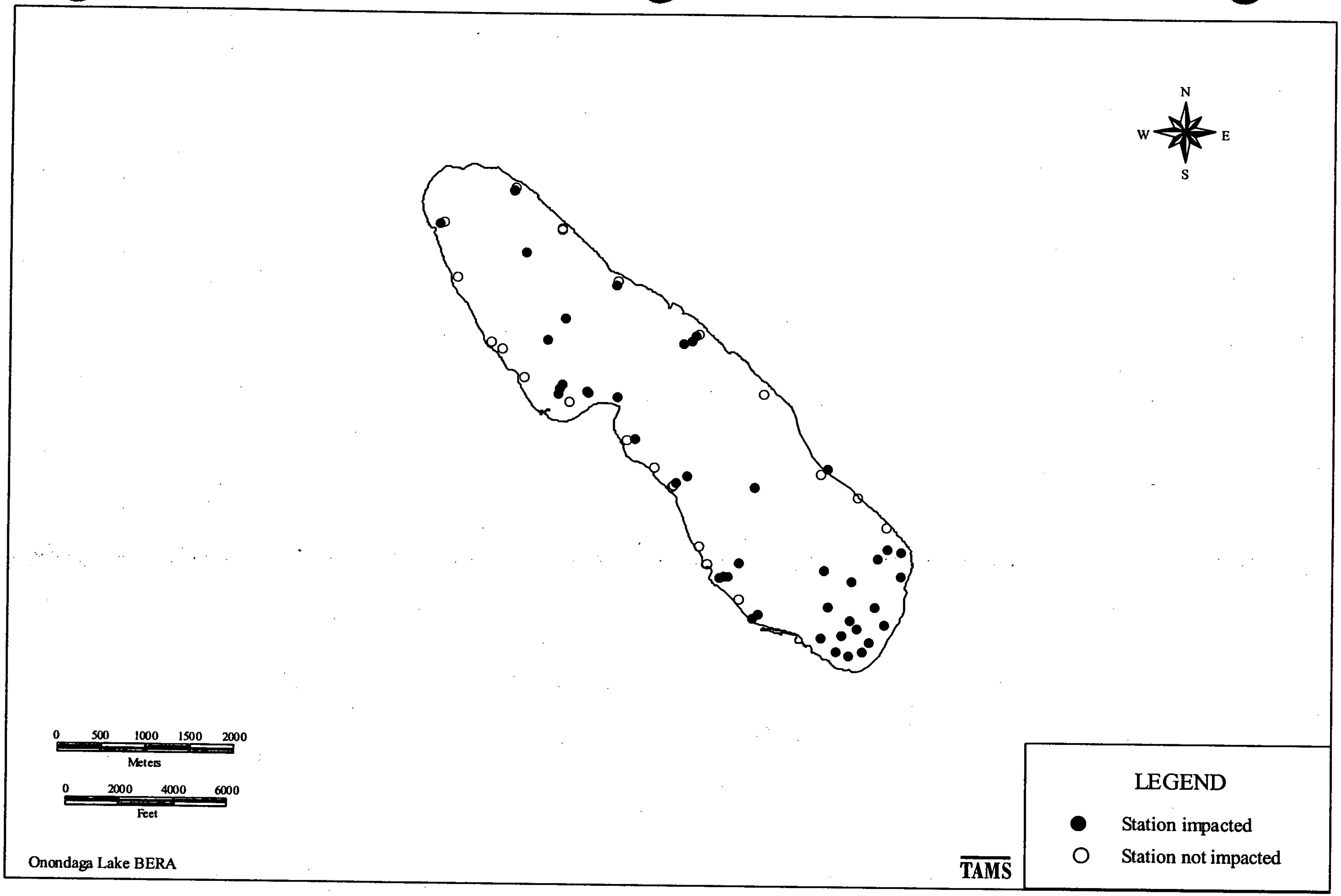


Figure 9-9
Patterns of Benthic Taxa Richness in Onondaga Lake in 1992

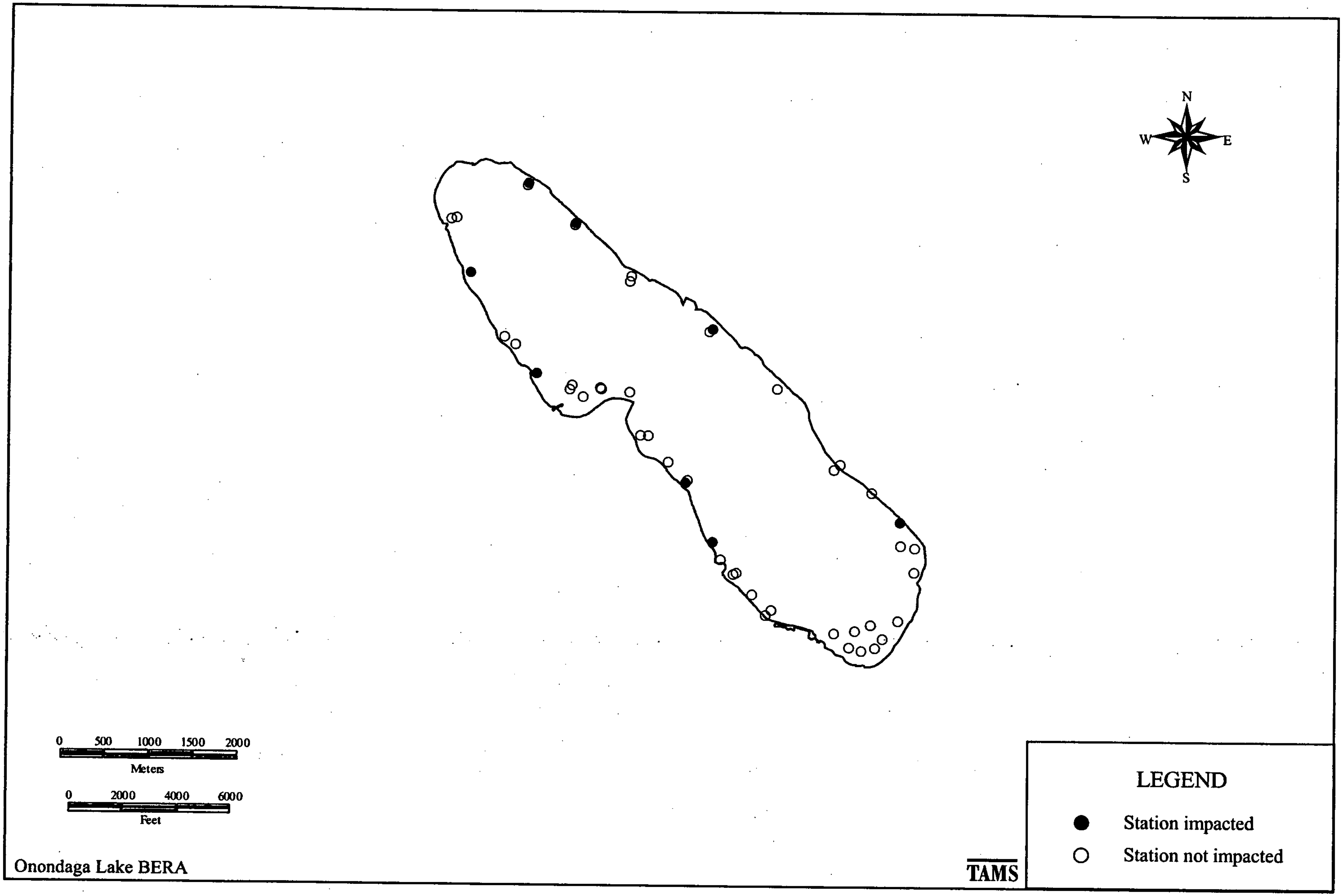


Figure 9-10
Patterns of Benthic Dominance in Onondaga Lake in 1992

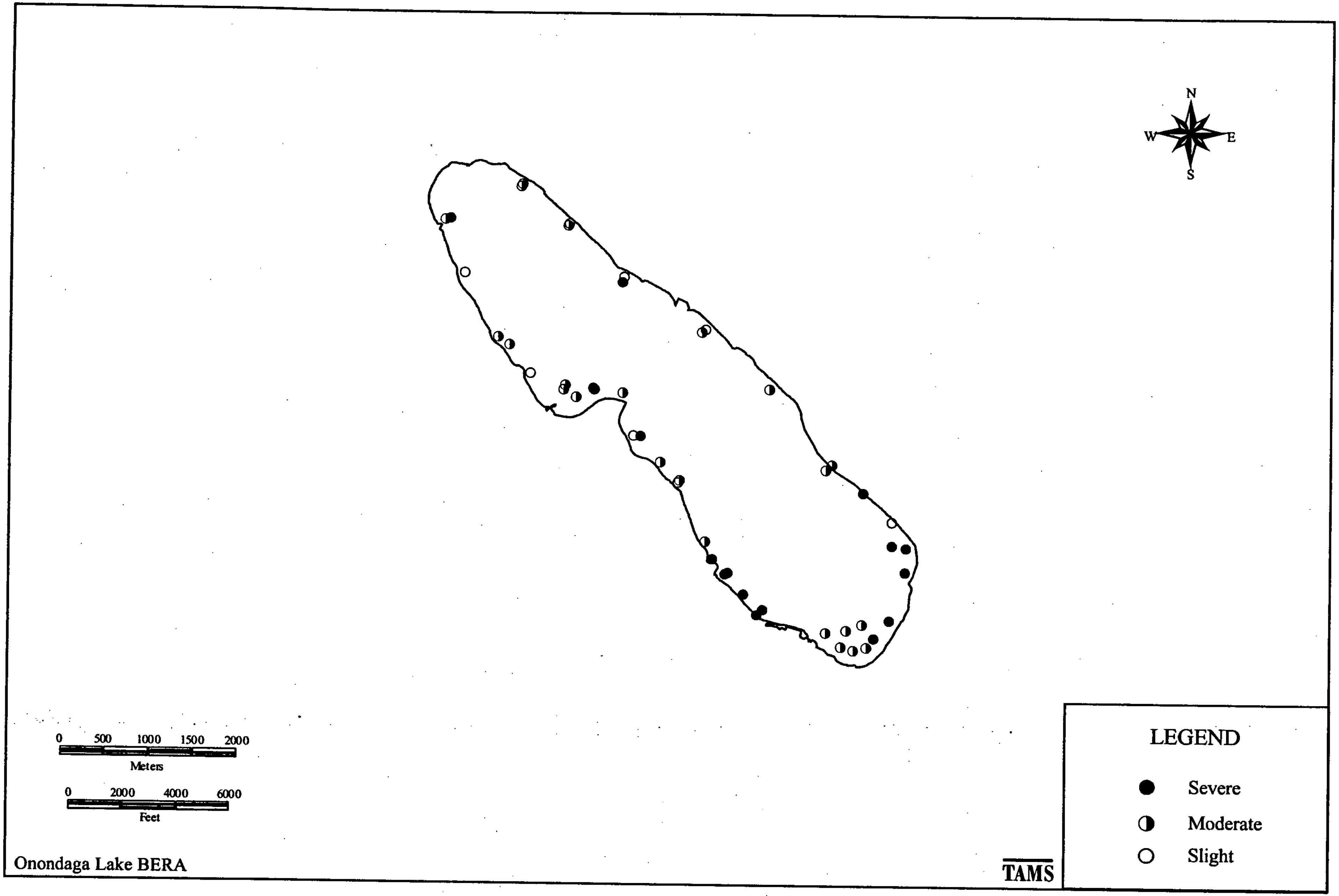


Figure 9-11

Patterns of Percent Model Affinity for Benthic Communities in Onondaga Lake in 1992

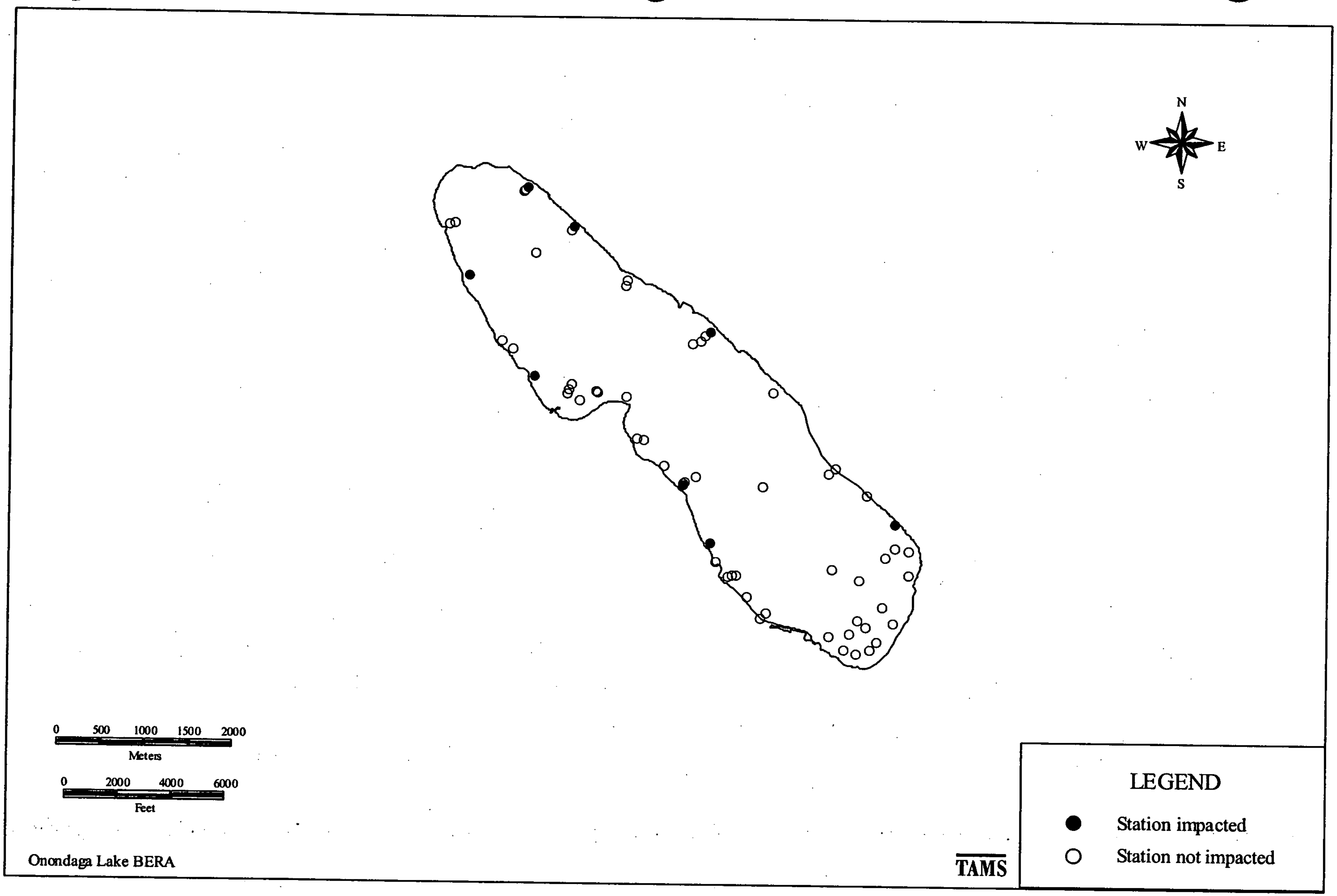
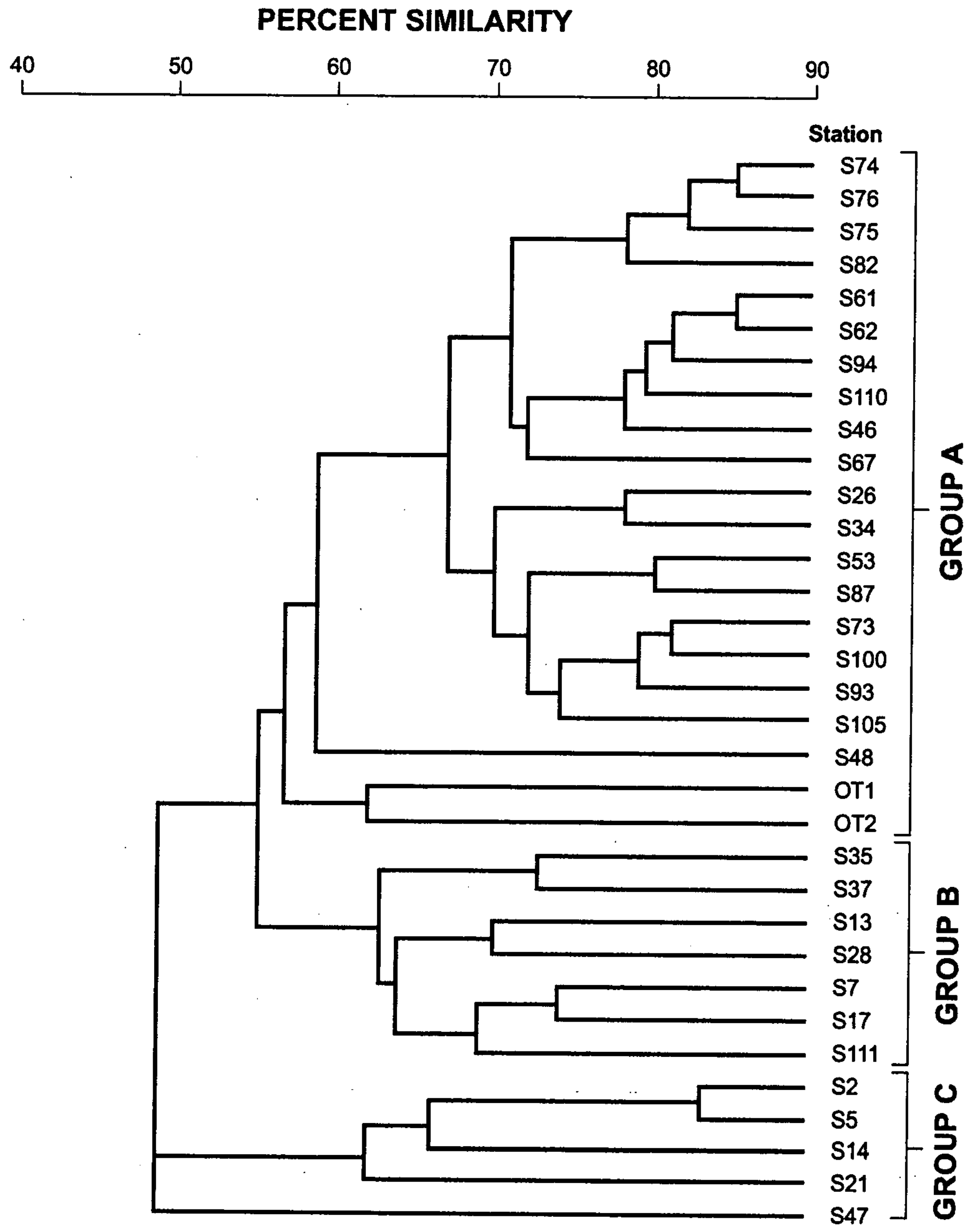


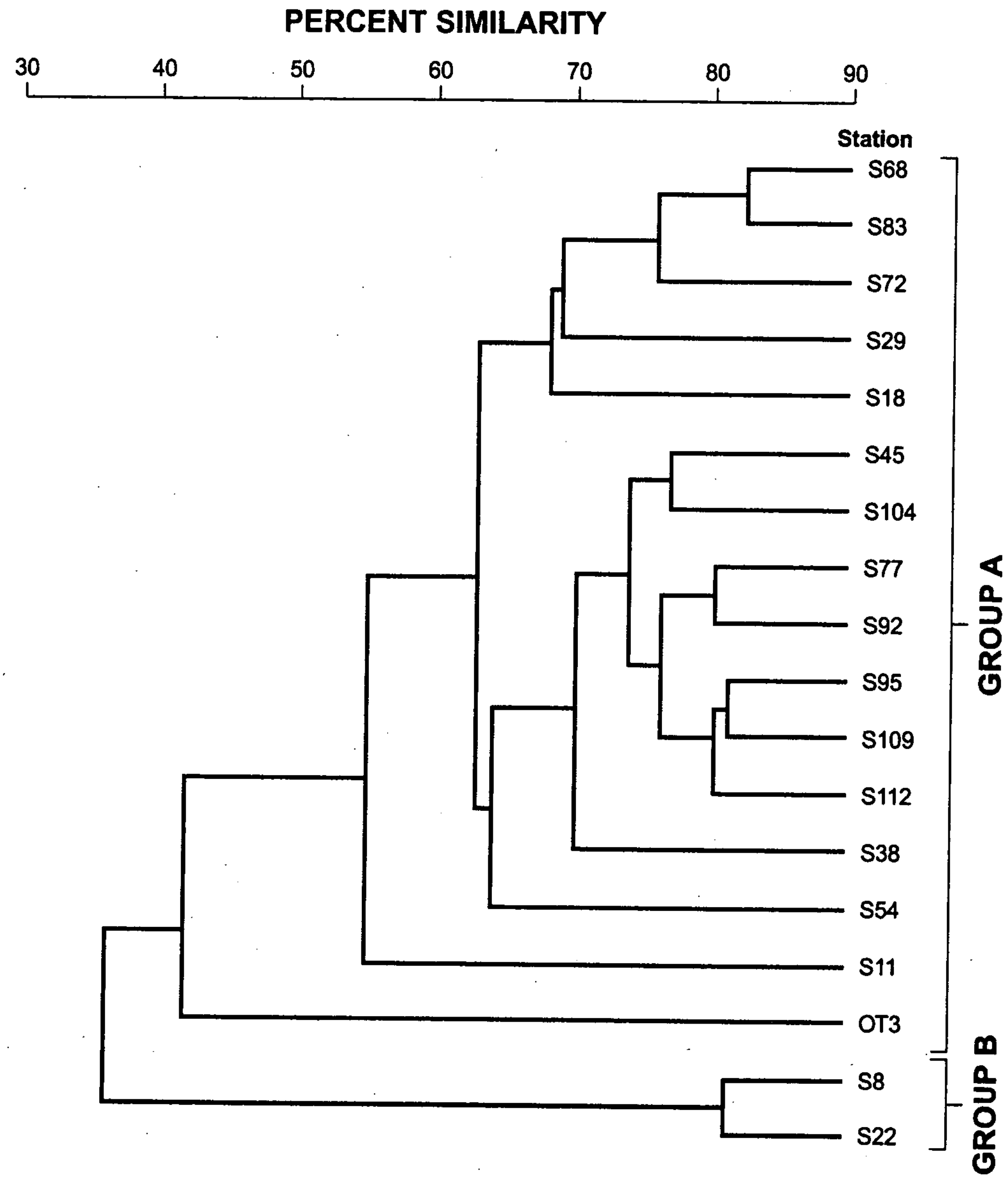
Figure 9-12
Patterns of Benthic Dominance in Onondaga Lake in 1992



Note: Classification analysis was based on the Bray-Curtis similarity index applied to log-transformed abundances of benthic macroinvertebrate taxa from stations in Onondaga (denoted as "S") and Otisco (denoted as "OT") lakes.

Source: Exponent, 2001b

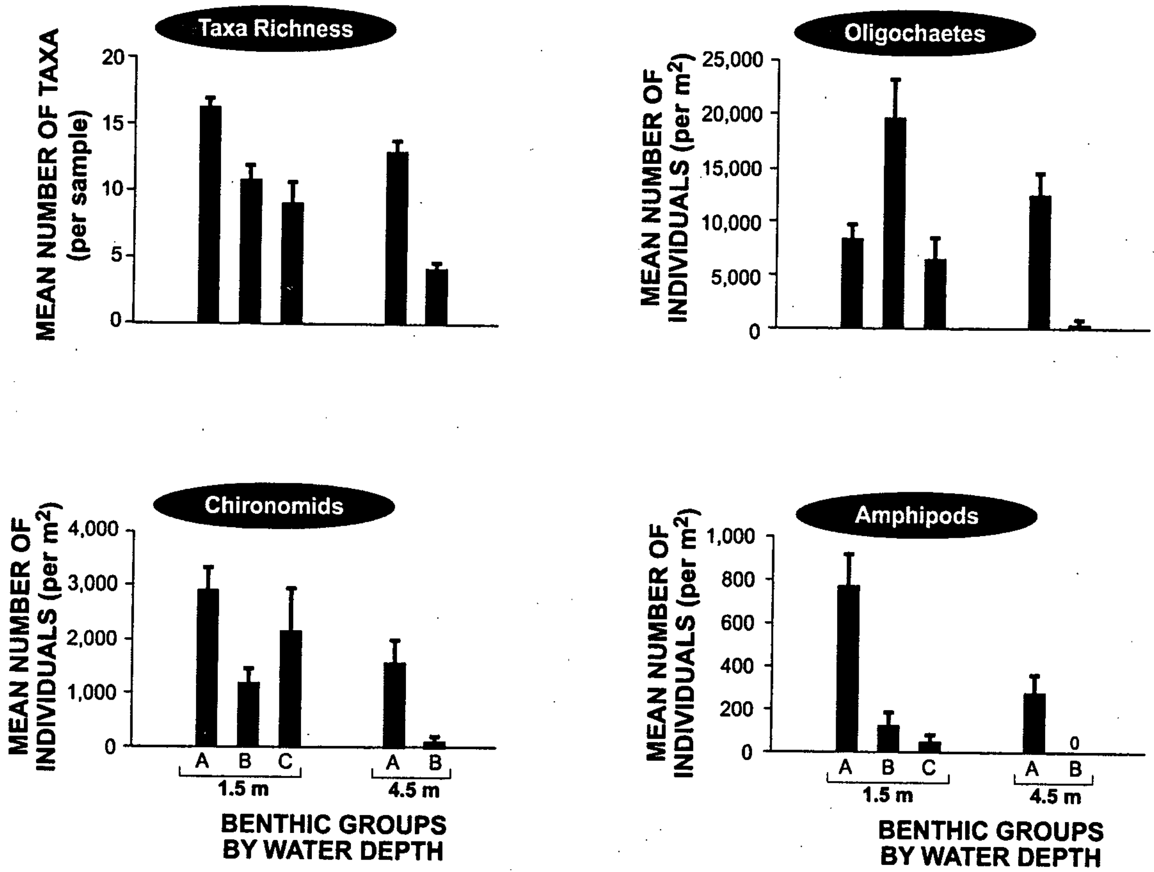
Figure 9-13. Results of classification analysis of benthic macroinvertebrate communities at 1.5-m stations in Onondaga and Otisco lakes in 1992



Note: Classification analysis was based on the Bray-Curtis similarity index applied to log-transformed abundances of benthic macroinvertebrate taxa from stations in Onondaga (denoted as "S") and Otisco (denoted as "OT") lakes.

Source: Exponent, 2001b

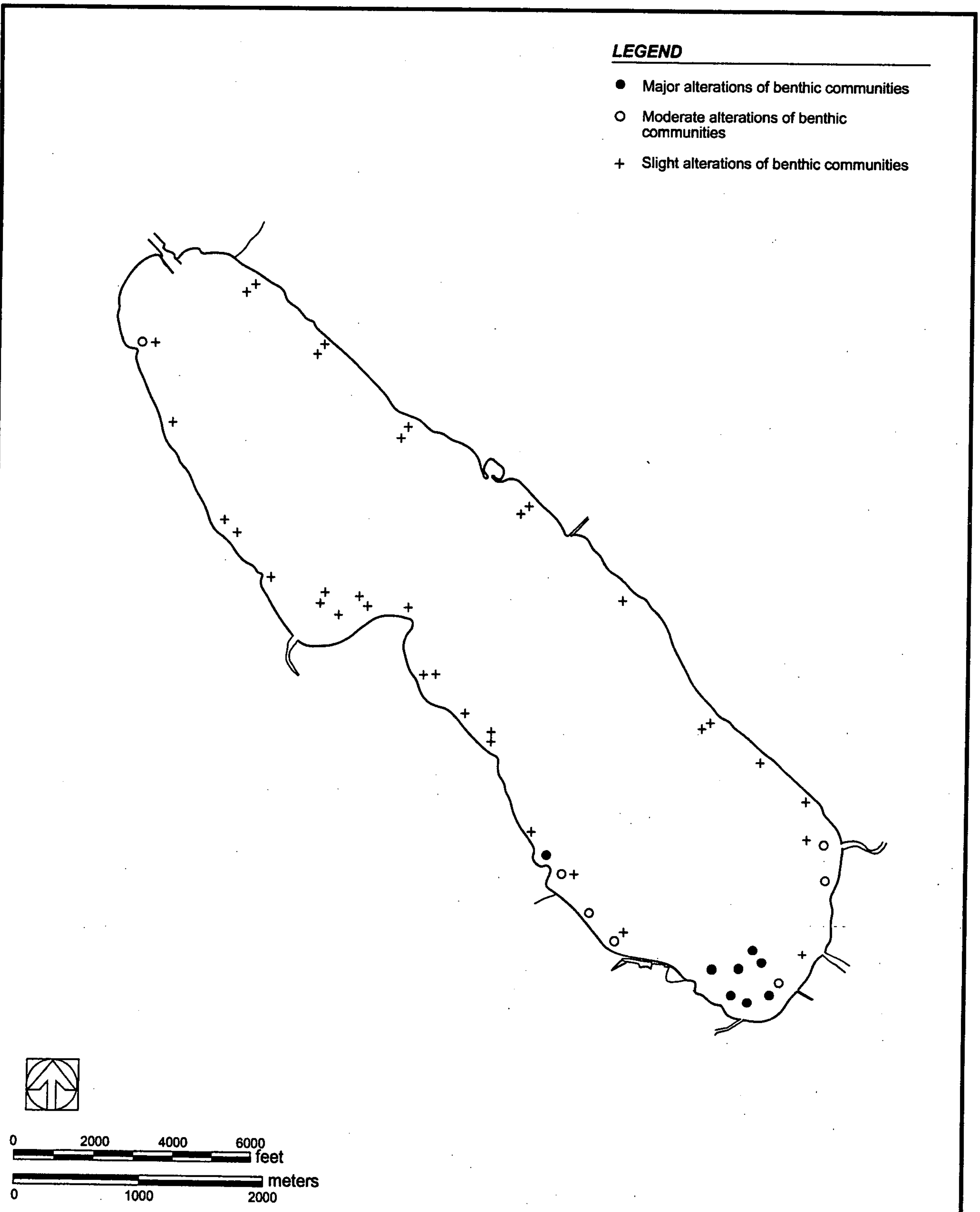
Figure 9-14. Results of classification analysis of benthic macroinvertebrate communities at 4.5-m stations in Onondaga and Otisco lakes in 1992



Note: Bars represent standard errors. Benthic groups are clusters of stations identified by classification analysis based on station-specific abundances of benthic macroinvertebrates.

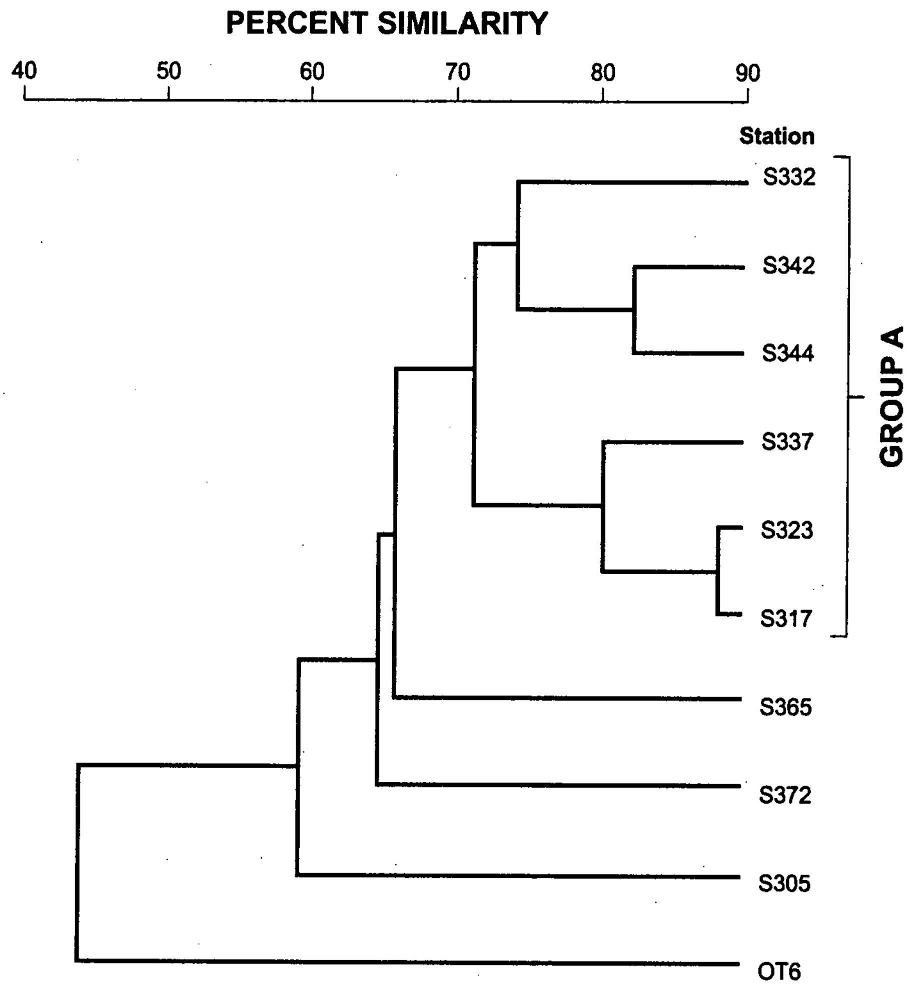
Source: Modified from Exponent, 2001b

Figure 9-15. Comparison of Major Benthic Macroinvertebrate Community Variables Among Benthic Groups for Onondaga Lake in 1992



Source: Exponent, 2001b

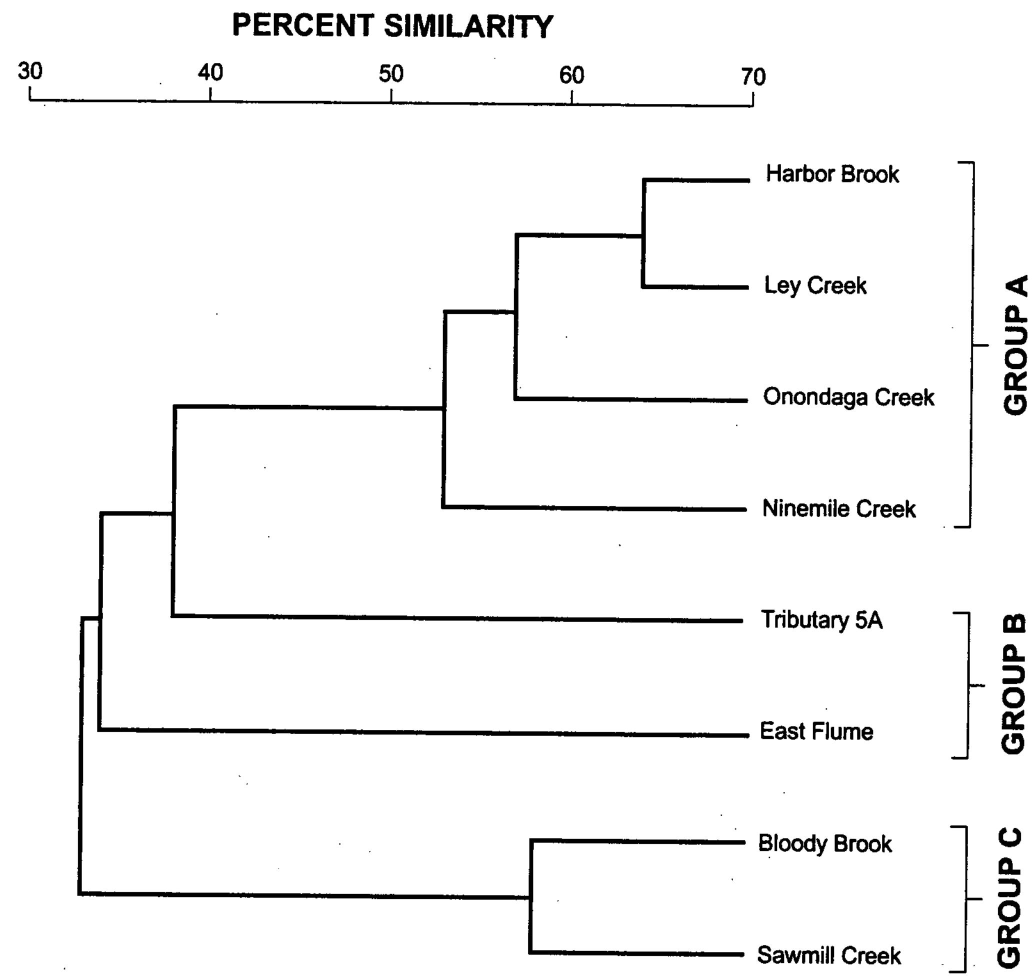
Figure 9-16. Locations of stations at which alterations of benthic macroinvertebrate communities were found in Onondaga Lake in 1992 based on classification analysis



Note: Classification analysis was based on the Bray-Curtis similarity index applied to log-transformed abundances of benthic macroinvertebrate taxa from stations in Onondaga (denoted as "S") and Otisco (denoted as "OT") lakes.

Source: Exponent, 2001b

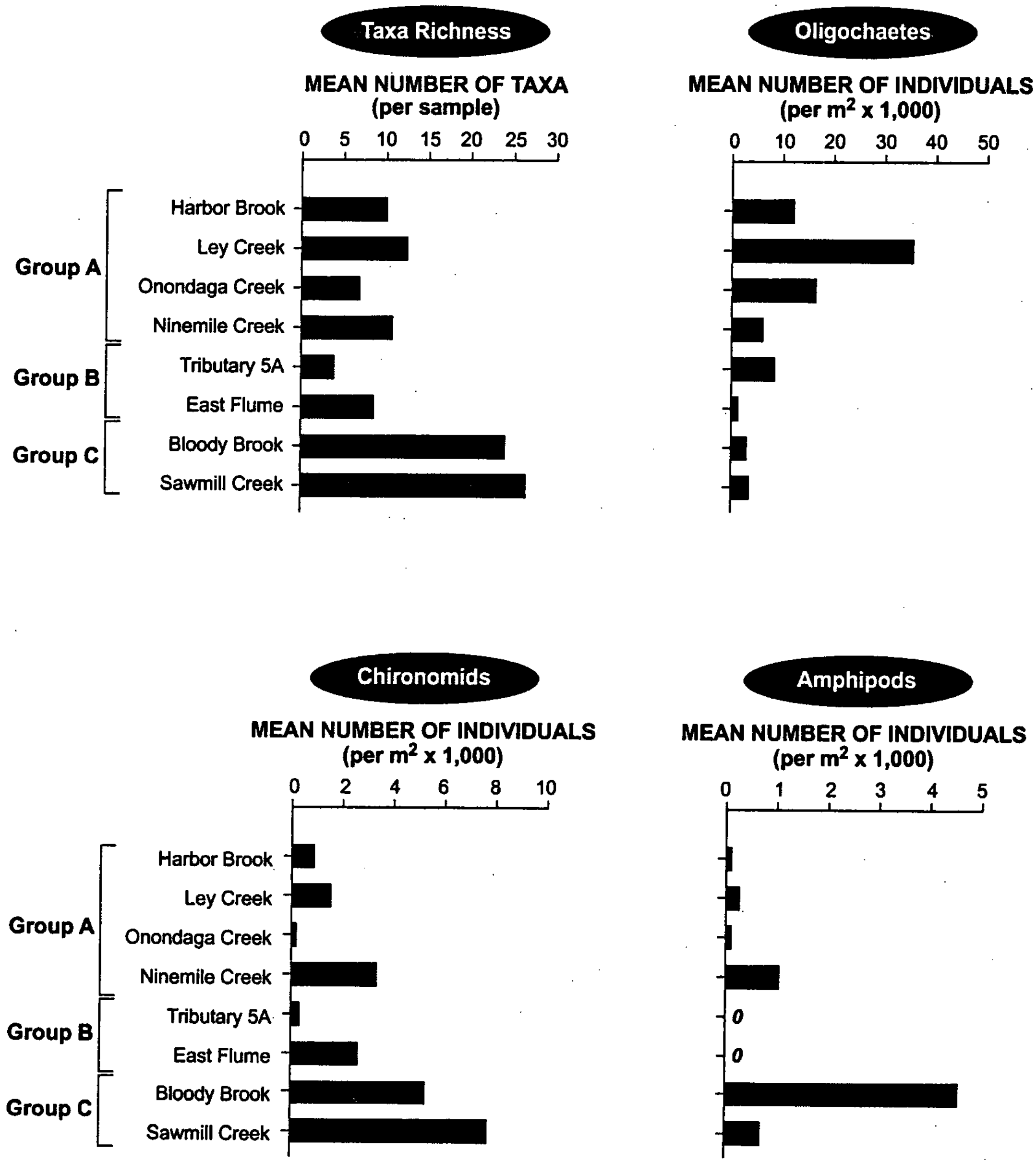
Figure 9-17. Results of classification analysis of benthic macroinvertebrate communities at shallow stations in Onondaga and Otisco lakes in 2000



Note: Classification analysis was based on the Bray-Curtis similarity index applied to log-transformed abundances of benthic macroinvertebrate taxa from each tributary.

Source: Exponent, 2001b

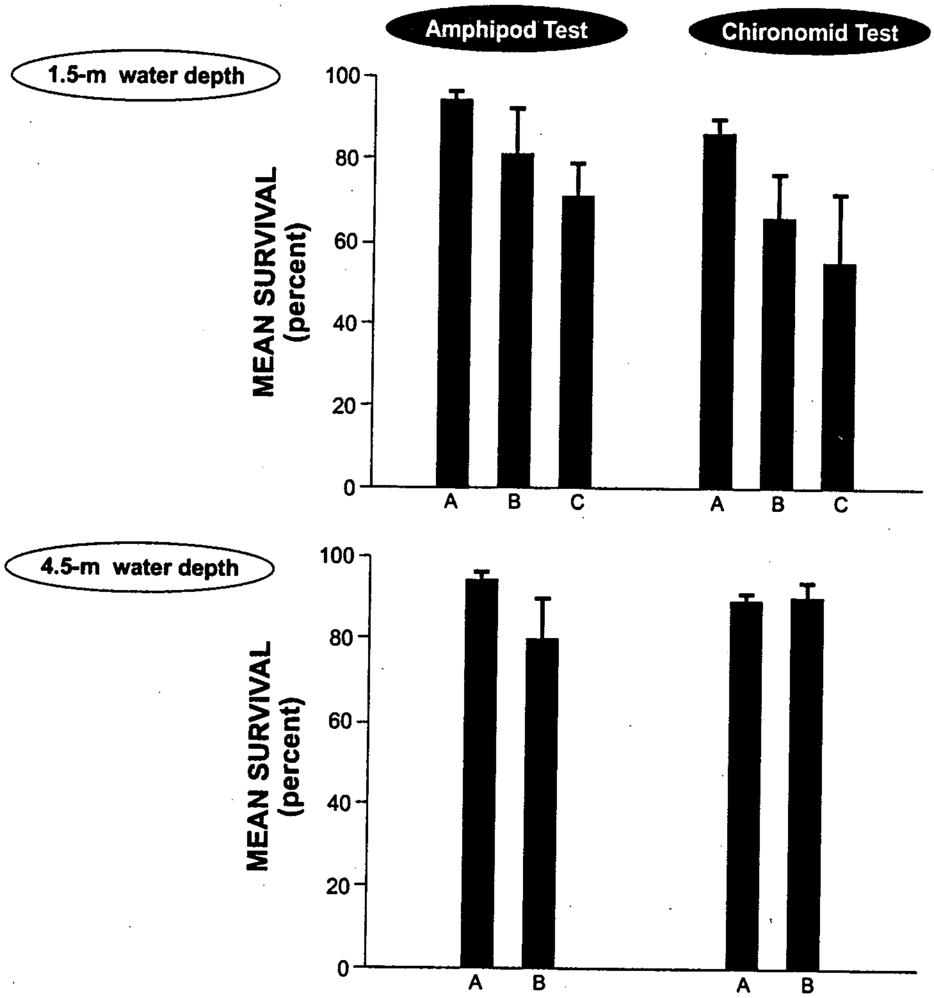
Figure 9-18. Results of classification analysis of benthic macroinvertebrate communities in tributaries of Onondaga Lake in 1992



Note: Benthic groups are clusters of stations identified by classification analysis based on abundance of benthic macroinvertebrates in each tributary.

Source: Exponent, 2001b

Figure 9-19. Comparison of major benthic macroinvertebrate community variables among tributaries of Onondaga Lake in 1992

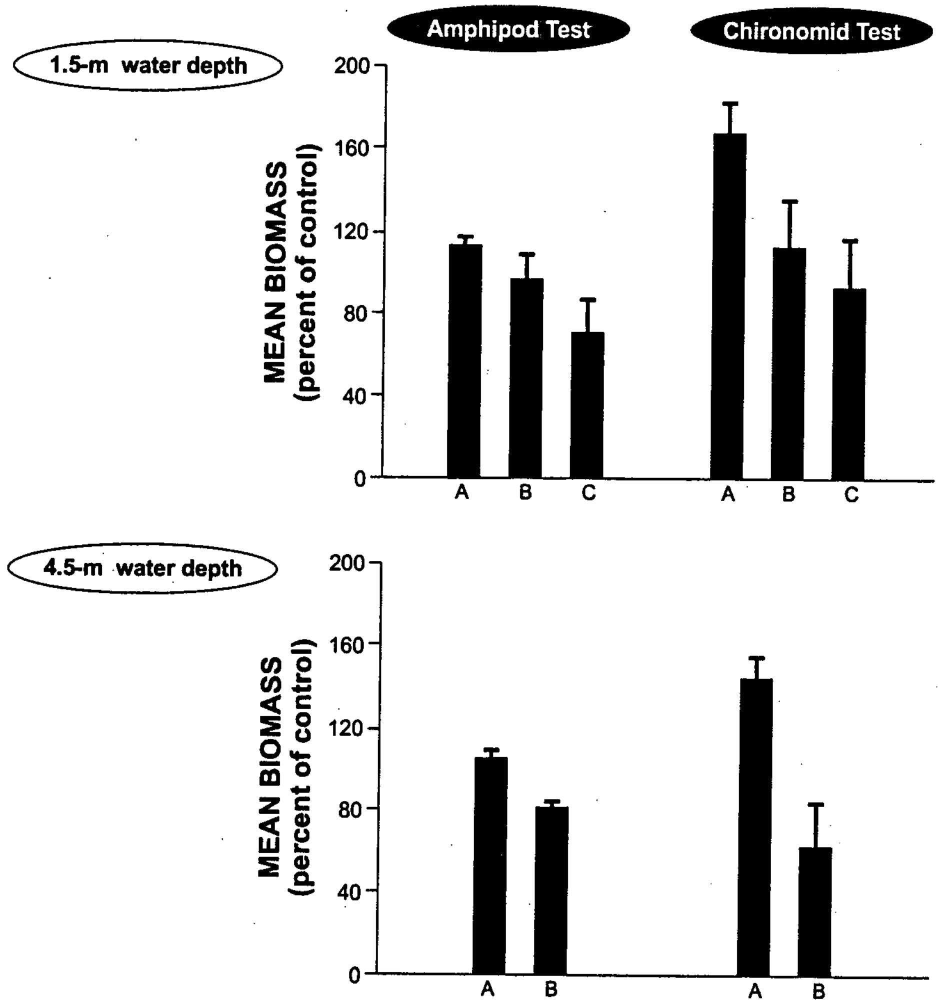


Note: Bars represent standard errors.

Benthic groups are clusters of stations identified by classification analysis based on station-specific abundances of benthic macroinvertebrates.

Source: Modified from Exponent, 2001b

Figure 9-20. Comparison of Survival Results for Toxicity Tests Among Benthic Macroinvertebrate Groups for Onondaga Lake in 1992



Note: Bars represent standard errors.
Benthic groups are clusters of stations identified by classification analysis based on station-specific abundances of benthic macroinvertebrates.

Source: Modified from Exponent, 2001b

Figure 9-21. Comparison of Biomass Results for Sediment Toxicity Tests Among Benthic Macroinvertebrate Groups for Onondaga Lake in 1992

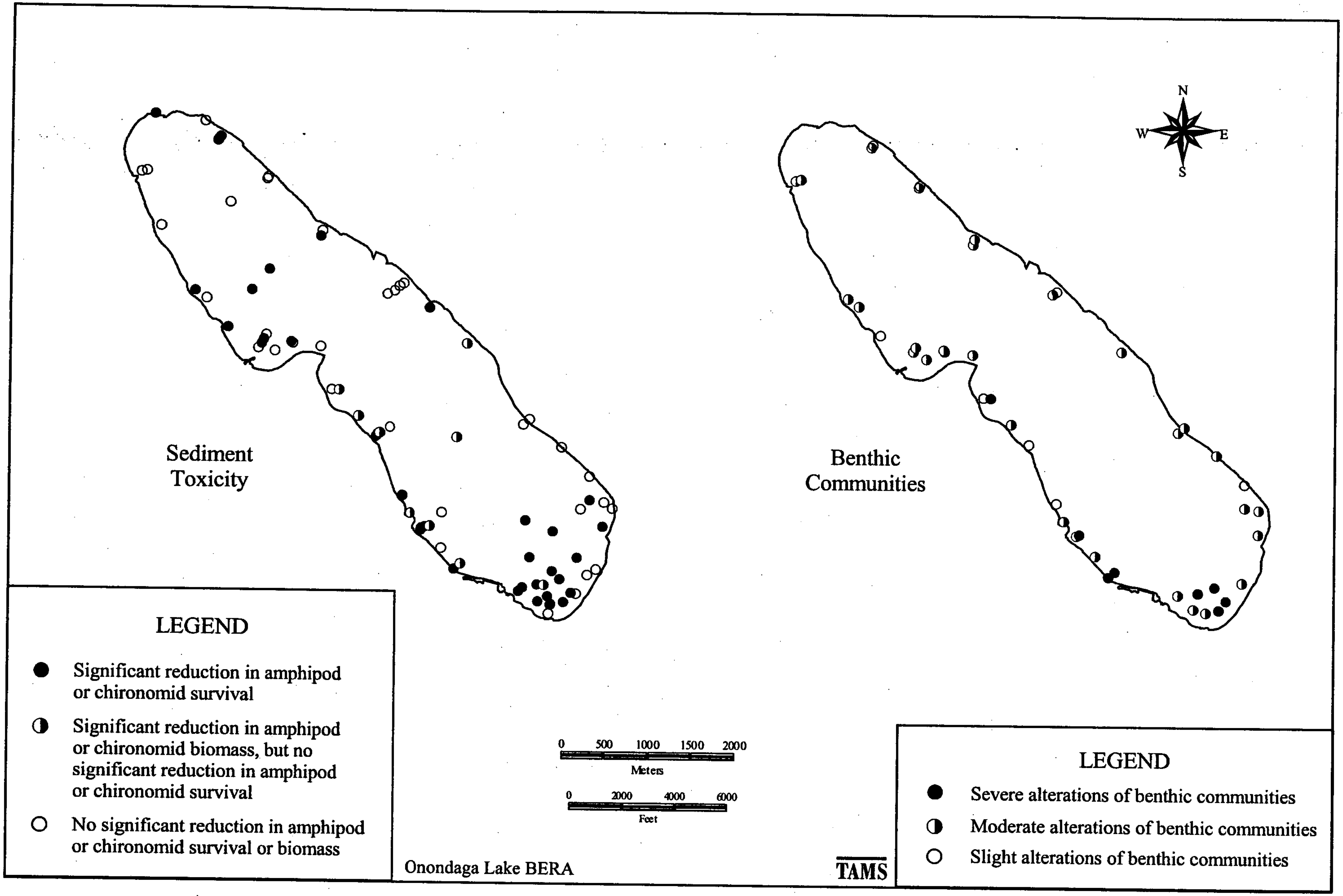


Figure 9-22
Locations of Stations at Which Significant Sediment Toxicity or Alterations of Benthic Macroinvertebrate Communities were Found in Onondaga Lake in 1992

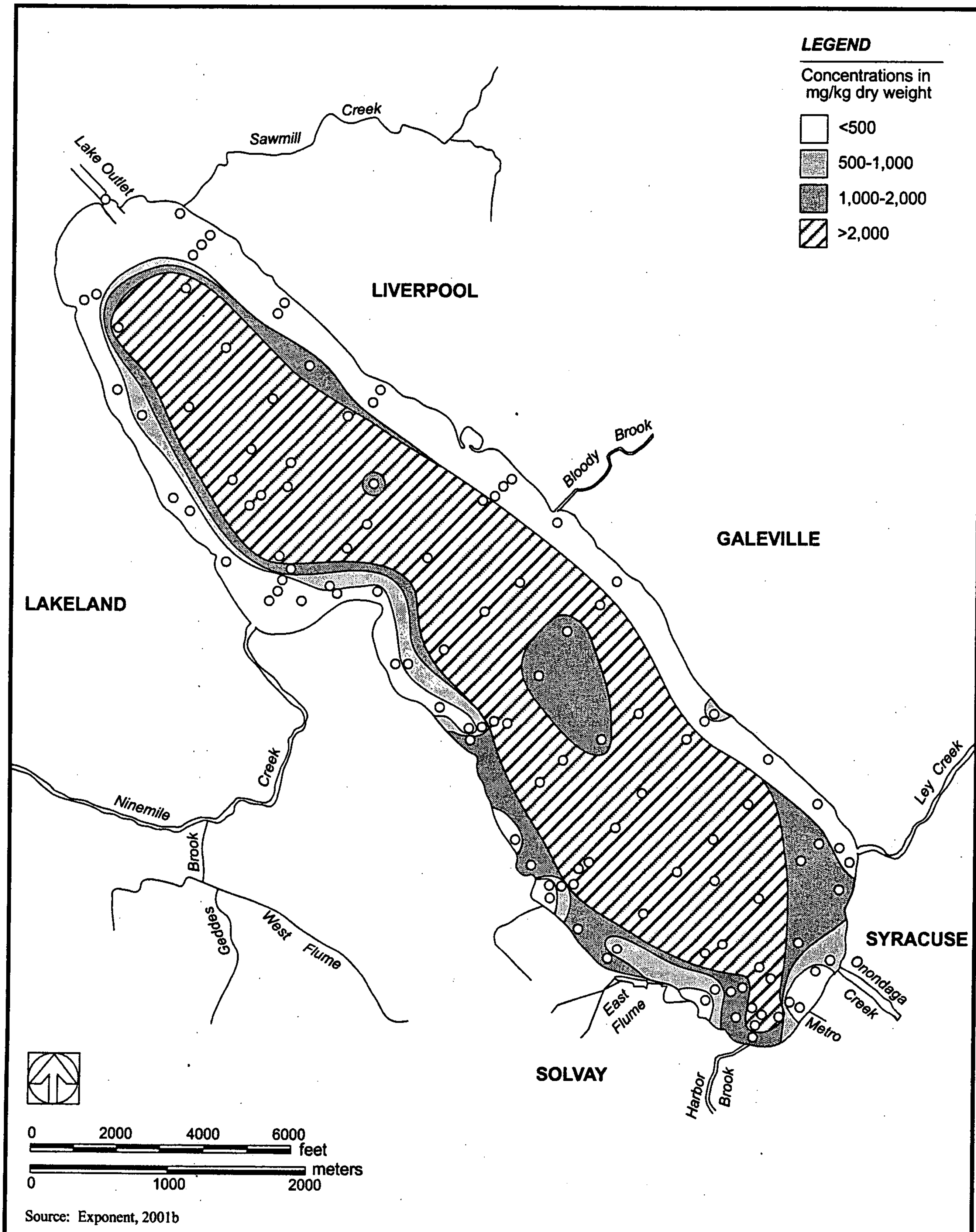


Figure 9-23. Distribution of acid-volatile sulfides in surficial sediments of Onondaga Lake in 1992.

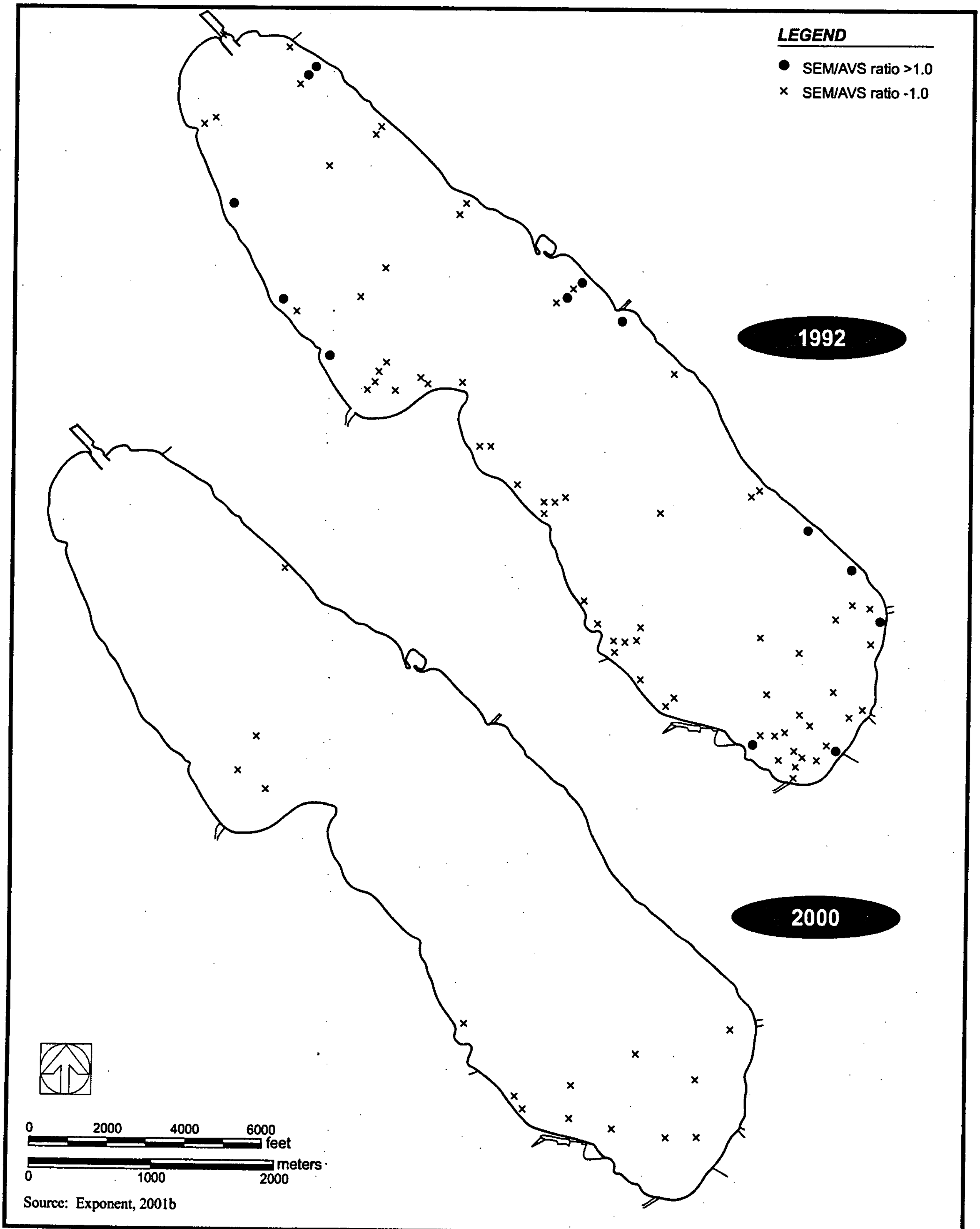


Figure 9-24. Locations of stations at which the SEM/AVS ratio exceeded 1.0 in surface sediment of Onondaga Lake in 1992 and 2000

Table 9-1. SEDQUAL Impacted/Non-impacted List for Stations Sampled in 1992

Station	Amphipod Biomass	Amphipod Survival	Chironomid Biomass	Chironomid Survival
S1	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S2	Impacted	Not Impacted	Not Impacted	Impacted
S3	Not Impacted	Not Impacted	Impacted	Impacted
S4	Not Impacted	Not Impacted	Impacted	Impacted
S5	Impacted	Not Impacted	Not Impacted	Impacted
S6	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S7	Not Impacted	Not Impacted	Not Impacted	Impacted
S8	Not Impacted	Not Impacted	Impacted	Impacted
S9	Impacted	Not Impacted	Impacted	Impacted
S10	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S11	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S12	Not Impacted	Not Impacted	Impacted	Impacted
S13	Not Impacted	Not Impacted	Not Impacted	Impacted
S14	Impacted	Not Impacted	Not Impacted	Impacted
S15	Impacted	Not Impacted	Impacted	Impacted
S16	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S17	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S18	Not Impacted	Not Impacted	Impacted	Impacted
S19	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S20	Impacted	Not Impacted	Impacted	Impacted
S21	Impacted	Not Impacted	Impacted	Impacted
S22	Impacted	Not Impacted	Not Impacted	Not Impacted
S24	Not Impacted	Not Impacted	Not Impacted	Impacted
S25	Not Impacted	Not Impacted	Not Impacted	Impacted
S26	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S27	Not Impacted	Not Impacted	Not Impacted	Impacted
S28	Impacted	Impacted	Impacted	Impacted
S29	Impacted	Not Impacted	Not Impacted	Not Impacted
S34	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S35	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S36	Impacted	Not Impacted	Impacted	Impacted
S37	Not Impacted	Not Impacted	Not Impacted	Impacted
S38	Impacted	Not Impacted	Impacted	Impacted
S39	Impacted	Not Impacted	Not Impacted	Not Impacted
S40	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S45	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S46	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S47	Impacted	Not Impacted	Not Impacted	Not Impacted
S48	Not Impacted	Not Impacted	Impacted	Impacted
S51	Not Impacted	Not Impacted	Impacted	Not Impacted
S53	Impacted	Not Impacted	Not Impacted	Impacted
S54	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S55	Impacted	Not Impacted	Not Impacted	Not Impacted
S56	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S61	Not Impacted	Not Impacted	Impacted	Not Impacted
S62	Impacted	Not Impacted	Not Impacted	Not Impacted
S66	Not Impacted	Not Impacted	Not Impacted	Impacted

Table 9-1. (cont.)

Station	Amphipod Biomass	Amphipod Survival	Chironomid Biomass	Chironomid Survival
S67	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S68	Impacted	Not Impacted	Not Impacted	Not Impacted
S70	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S71	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S72	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S73	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S74	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S75	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S76	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S77	Not Impacted	Not Impacted	Not Impacted	Impacted
S81	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S82	Not Impacted	Not Impacted	Not Impacted	Impacted
S83	Not Impacted	Not Impacted	Not Impacted	Impacted
S84	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S86	Not Impacted	Not Impacted	Not Impacted	Impacted
S87	Not Impacted	Not Impacted	Not Impacted	Impacted
S90	Not Impacted	Not Impacted	Not Impacted	Impacted
S92	Not Impacted	Not Impacted	Not Impacted	Impacted
S93	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S94	Not Impacted	Not Impacted	Not Impacted	Impacted
S95	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S100	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S103	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S104	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S105	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S108	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S109	Not Impacted	Not Impacted	Not Impacted	Impacted
S110	Not Impacted	Not Impacted	Not Impacted	Impacted
S111	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S112	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S113	Not Impacted	Not Impacted	Not Impacted	Not Impacted
S114	Not Impacted	Not Impacted	Not Impacted	Impacted

Notes: Impacted = the station is a hit for this parameter (impacted).
Not Impacted = the station is not a hit for this parameter (non-impacted).

Table 9-2. SEDQUAL Impacted/Non-impacted List for Stations Sampled in 2000

Station	Amphipod Biomass	Amphipod Survival	Amphipod Reproduction	Chironomid Biomass	Chironomid Survival	Chironomid Emergence
S302	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Impacted	Non-Impacted
S303	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Impacted	Non-Impacted
S305	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted
S315	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted
S317	Impacted	Impacted	Non-Impacted	Impacted	Non-Impacted	Non-Impacted
S320	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Impacted	Non-Impacted
S323	Impacted	Non-Impacted	Non-Impacted	Impacted	Impacted	Non-Impacted
S332	Non-Impacted	Impacted	Non-Impacted	Impacted	Impacted	Impacted
S337	Non-Impacted	Impacted	Non-Impacted	Impacted	Impacted	Impacted
S342	Non-Impacted	Impacted	Impacted	Impacted	Impacted	Impacted
S344	Non-Impacted	Impacted	Impacted	Impacted	Impacted	Impacted
S354	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Impacted
S355	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted	Non-Impacted
S365	Non-Impacted	Impacted	Impacted	Impacted	Impacted	Non-Impacted
S372	Non-Impacted	Non-Impacted	Non-Impacted	Impacted	Non-Impacted	Non-Impacted

Notes: Impacted = the station is a hit for this parameter (impacted).
Not Impacted = the station is not a hit for this parameter (non-impacted).

Table 9-3. Onondaga Lake Benthic Analysis Assessment Criteria - Ranges

Criterion	Non-Impaired	Slightly Impaired	Moderately Impaired	Severely Impaired
Total Species Richness at Station [1]	>32	25 - 32	14 - 24	0 - 13
Dominance Index [1]	<61	61 - 75	76 - 90	91 - 100
Total # of NCO [1]	>15	10 - 15	5 - 9	0 - 4
Community Composition (Percent Model Affinity) [2]	>64	50 - 64	35 - 49	<35
Shannon-Wiener [3] (Diversity Index)	3.1 - 4.0+	2.1 - 3.0	1.5 - 2.0	<1.5

Notes: Cumulative review of the five metrics is used to make an overall impairment assessment of a station.
Sources:
(1) NYSDEC (letter dated May 27, 1999).
(2) Bode et al. (1991).
(3) Based on USEPA (1973).

Table 9-4. Onondaga Lake Benthic Community Analysis, 1992 Data

Station	Water Depth (m)	Total # of Taxa	Taxa Richness (avg of 5 Reps)	Dominance Index	Total # of NCO	NCO (avg. of 5 reps.)	Community Composition (PMA)	Shannon-Wiener (Diversity Index)
S2	1.5	9	7	82%	2	<1	40	2
S5	1.5	12	8	83%	0	0	40	<1
S7	1.5	10	6	93%	2	<1	30	1.5
S8	4.5	8	5	92%	0	0	36	1.3
S11	4.5	19	13	85%	6	2	29	1.7
S13	1.5	17	10	93%	6	2	26	1.6
S14	1.5	9	6	90%	1	<1	39	1.6
S17	1.5	21	12	90%	8	4	27	1.3
S18	4.5	17	9	89%	6	1	27	1.5
S21	1.5	16	8	92%	9	4	39	1.7
S22	4.5	7	4	96%	0	0	40	1.6
S26	1.5	25	16	71%	8	5	56	2.6
S28	1.5	13	6	97%	3	1	21	1.3
S29	4.5	11	7	84%	3	1	30	2
S34	1.5	23	17	80%	7	5	27	2.2
S35	1.5	29	13	91%	14	4	27	1.6
S37	1.5	28	14	92%	13	5	28	1.5
S38	4.5	20	11	93%	7	5	26	1.2
S45	4.5	27	15	82%	11	6	40	1.9
S46	1.5	23	14	78%	6	3	40	2.2
S47	1.5	30	15	90%	15	6	30	1.4
S48	1.5	28	17	59%	14	7	44	3.1
S53	1.5	25	16	63%	11	5	41	3
S54	4.5	35	15	79%	17	5	44	2.1
S61	1.5	24	16	80%	6	3	40	2.1
S62	1.5	27	16	80%	9	4	41	4.3
S67	1.5	23	13	76%	10	4	52	2.3
S68	4.5	16	9	88%	3	1	34	1.4
S72	4.5	15	10	91%	3	1	39	1.5
S73	1.5	30	22	72%	11	7	58	2.7
S74	1.5	18	11	81%	4	2	36	2
S75	1.5	22	16	69%	7	3	42	2
S76	1.5	22	13	61%	7	3	50	2.1
S77	4.5	16	12	85%	5	3	31	1.9

Table 9-4. (cont.)

Station	Water Depth (m)	Total # of Taxa	Taxa Richness (avg of 5 Reps)	Dominance Index	Total # of NCO	NCO (avg. of 5 reps.)	Community Composition (PMA)	Shannon-Wiener (Diversity Index)
S82	1.5	17	9	94%	5	2	41	1.4
S83	4.5	20	10	92%	6	2	39	1.5
S87	1.5	27	19	61%	11	6	54	3
S92	4.5	23	14	89%	6	4	29	1.6
S93	1.5	23	18	78%	5	5	50	2.1
S94	1.5	25	16	80%	8	4	39	2
S95	4.5	25	17	80%	7	4	42	2.4
S100	1.5	28	21	50%	8	7	60	3.6
S104	4.5	24	13	75%	8	6	48	2.3
S105	1.5	28	21	64%	11	7	59	3
S109	4.5	24	16	78%	8	4	39	2.2
S110	1.5	28	20	64%	10	5	47	3
S111	1.5	22	13	80%	6	2	38	2.1
S112	4.5	27	17	85%	10	5	34	1.8
OT1	1.5	28	14	88%	10	4	40	1.6
OT2	1.5	32	15	87%	15	5	37	1.5
OT3	4.5	33	19	81%	22	11	50	2.3
T1	0.5	25	10	94%	14	4	-----	1.2
T3	4.5	11	6	94%	3	<1	-----	1.5
T5	1	25	12	95%	11	4	-----	1
T7	0.5	18	8	80%	3	2	-----	2
T9	0.5	9	4	99%	4	1	-----	1.1
T11	0.5	41	23	58%	20	9	-----	3.4
T13	2	20	11	87%	9	4	-----	1.9
T15	0.5	43	26	53%	27	13	-----	3.5

Notes:

1. Other than "Total # of Taxa" and "Total # of NCO", data presented are based on averaging the five replicates and not pooling the numbers of all replicates.
2. For Community Composition, each Tributary station is designated as "-----" - cannot compare tributary stations to a model lake community.
3. Water depths obtained from PTI (1993). Stations deeper than 4.5 m not included (see text).
4. NCO = Non-Chironomid/Oligochaete
   PMA = Percent Model Affinity

Table 9-5. Onondaga Lake and Otisco Lake Benthic Stations for Evaluation - Impairment Assessment, 1992 Data

Station	Total # of Taxa	Total # of NCO	Dominance Index	Community Composition (PMA)	Shannon- Wiener (Diversity Index)	Impairment Assessment
S2	9 = I	82% = M	2 = I	40 = M	2.0 = M	Moderately impaired
S5	12 = I	83% = M	0 = I	40 = M	<1 = I	Severely impaired
S7	10 = I	93% = I	2 = I	30 = I	1.5 = M	Severely impaired
S8	8 = I	92% = I	0 = I	36 = M	1.3 = I	Severely impaired
S11	19 = M	85% = M	6 = M	29 = I	1.7 = M	Moderately impaired
S13	17 =M	93% = I	6 = M	26 = I	1.6 = M	Moderately impaired
S14	9 = I	90% = M	1 = I	39 = M	1.6 = M	Moderately impaired
S17	21 = M	90% = M	8 = M	27 = I	1.3 = I	Moderately impaired
S18	17 = M	89% = M	6 = M	27 = I	1.5 = M	Moderately impaired
S21	16 = M	92% = I	9 = M	39 = M	1.7 = M	Moderately impaired
S22	7 = I	96% = I	0 = I	40 = M	1.6 = M	Severely impaired
S26	25 = S	71% = S	8 = M	56 = S	2.6 = S	Slightly impaired
S28	13 = I	97% = I	3 = I	21 = I	1.3 = I	Severely impaired
S29	11 = I	84% = M	3 = I	30 = I	2.0 = M	Severely impaired
S34	23 = M	80% = M	7 = M	27 = I	2.2 = S	Moderately impaired
S35	29 = S	91% = I	14 = S	27 = I	1.6 = M	Moderately impaired*
S37	28 = S	92% = I	13 = S	28 = I	1.5 = M	Moderately impaired*
S38	20 = M	93% = I	7 = M	26 = I	1.2 = I	Severely impaired
S45	27 = S	82% = M	11 = S	40 = M	1.9 = M	Moderately impaired
S46	23 = M	78% = M	6 = M	40 = M	2.2 = S	Moderately impaired
S47	30 = S	90% = M	15 = S	30 = I	1.4 = I	Moderately impaired*
S48	28 = S	59% = S	14 = S	44 = M	3.1 = N	Slightly impaired
S53	25 = S	63% = S	11 = S	41 = S	3.0 = S	Slightly impaired
S54	35 = N	79% = M	17 = N	44 = M	2.1 = S	Slightly impaired*
S61	24 = M	80% = M	6 = M	40 = M	2.1 = S	Moderately impaired
S62	27 = S	80% = M	9 = M	41 = M	4.3 = S	Moderately impaired
S67	23 = S	76% = M	10 = S	52 = S	2.3 = S	Slightly impaired
S68	16 = M	88% = M	3 = I	34 = I	1.4 = I	Severely impaired
S72	15 = M	91% = I	3 = I	39 = M	1.5 = M	Moderately impaired
S73	30 = S	72% = S	11 = S	58 = S	2.7 = S	Slightly impaired
S74	18 = M	81% = M	4 = I	36 = M	2.0 = M	Moderately impaired
S75	22 = M	69% = S	7 = M	42 = M	2.0 = M	Moderately impaired
S76	22 = M	61% = S	7 = M	50 = S	2.1 = S	Slightly impaired
S77	16 = M	85% = M	5 = M	31 = I	1.9 = M	Moderately impaired
S82	17 = M	94% = I	5 = M	41 = M	1.4 = I	Moderately impaired

Table 9-5. (cont.)

Station	Total # of Taxa	Total # of NCO	Dominance Index	Community Composition (PMA)	Shannon- Wiener (Diversity Index)	Impairment Assessment
S83	20 = M	92% = I	6 = M	39 = M	1.5 = M	Moderately impaired
S87	27 = S	61% = S	11 = S	54 = S	3.0 = S	Slightly impaired
S92	23 = M	89% = M	6 = M	29 = I	1.6 = M	Moderately impaired
S93	23 = M	78% = M	5 = M	50 = S	2.1 = S	Moderately impaired
S94	25 = S	80% = M	8 = M	39 = M	2.0 = M	Moderately impaired
S95	25 = S	80% = M	7 = M	42 = M	2.4 = S	Moderately impaired
S100	28 = S	50% = N	8 = M	60 = S	3.6 = N	Slightly impaired*
S104	24 = M	75% = S	8 = M	48 = M	2.3 = S	Moderately impaired
S105	28 = S	64% = S	11 = S	59 = S	3.0 = S	Slightly impaired
S109	24 = M	78% = M	8 = M	39 = M	2.2 = S	Moderately impaired
S110	28 = S	64% = S	10 = S	47 = M	3.0 = S	Slightly impaired
S111	22 = M	80% = M	6 = M	38 = M	2.1 = S	Moderately impaired
S112	27 = S	85% = M	10 = S	34 = I	1.8 = M	Moderately impaired*
OT1	28 = S	88% = M	10 = S	40 = M	1.6 = M	Moderately impaired
OT2	32 = S	87% = M	15 = S	37 = M	1.5 = M	Moderately impaired
OT3	33 = N	81% = M	22 = N	50 = S	2.3 = S	Slightly impaired*

Notes:

1. N = Non-impaired; S = Slightly impaired; M = Moderately impaired; I = Severely impaired
2. Overall station assessment on the basis that three or more of the five metrics exhibited the same impairment category.
3. Stations deeper than 4.5 m not included (see text).
4. NCO = Non-Chironomid/Oligochaete
   PMA = Percent Model Affinity

* Three or more of the five metrics did not exhibit the same impairment category; professional judgment and the results of all five metrics used for impairment assessment.

Table 9-6. Onondaga Lake and Otisco Lake Benthic Stations Statistical Evaluation, 1992 Data

Station	Total # of Taxa	Total # of NCO	Dominance Index	Shannon-Wiener (Diversity Index)
S2	Impacted	Impacted	Not Impacted	Impacted
S5	Impacted	Impacted	Not Impacted	Impacted
S7	Impacted	Impacted	Not Impacted	Impacted
S8	Impacted	Impacted	Not Impacted	Impacted
S11	Impacted	Impacted	Not Impacted	Impacted
S13	Impacted	Impacted	Not Impacted	Impacted
S14	Impacted	Impacted	Not Impacted	Impacted
S17	Impacted	Impacted	Not Impacted	Impacted
S18	Impacted	Impacted	Not Impacted	Impacted
S21	Impacted	Impacted	Not Impacted	Impacted
S22	Impacted	Impacted	Not Impacted	Impacted
S26	Not Impacted	Impacted	Impacted	Not Impacted
S28	Impacted	Impacted	Not Impacted	Impacted
S29	Impacted	Impacted	Not Impacted	Impacted
S34	Not Impacted	Impacted	Not Impacted	Not Impacted
S35	Not Impacted	Impacted	Not Impacted	Impacted
S37	Impacted	Impacted	Not Impacted	Impacted
S38	Impacted	Impacted	Not Impacted	Impacted
S45	Not Impacted	Impacted	Not Impacted	Impacted
S46	Impacted	Impacted	Not Impacted	Not Impacted
S47	Not Impacted	Impacted	Not Impacted	Impacted
S48	Not Impacted	Impacted	Impacted	Not Impacted
S53	Not Impacted	Impacted	Impacted	Not Impacted
S54	Not Impacted	Impacted	Not Impacted	Impacted
S61	Not Impacted	Impacted	Not Impacted	Not Impacted
S62	Not Impacted	Impacted	Not Impacted	Not Impacted
S67	Not Impacted	Impacted	Not Impacted	Not Impacted
S68	Impacted	Impacted	Not Impacted	Impacted
S72	Impacted	Impacted	Not Impacted	Impacted
S73	Not Impacted	Impacted	Impacted	Not Impacted
S74	Impacted	Impacted	Not Impacted	Not Impacted
S75	Not Impacted	Impacted	Not Impacted	Impacted
S76	Impacted	Impacted	Not Impacted	Not Impacted
S77	Impacted	Impacted	Not Impacted	Impacted
S82	Impacted	Impacted	Not Impacted	Impacted
S83	Impacted	Impacted	Not Impacted	Impacted
S87	Not Impacted	Impacted	Impacted	Not Impacted
S92	Impacted	Impacted	Not Impacted	Impacted
S93	Not Impacted	Impacted	Not Impacted	Not Impacted
S94	Not Impacted	Impacted	Not Impacted	Not Impacted
S95	Not Impacted	Impacted	Not Impacted	Not Impacted
S100	Not Impacted	Impacted	Impacted	Impacted
S104	Not Impacted	Impacted	Not Impacted	Not Impacted
S105	Not Impacted	Impacted	Impacted	Not Impacted
S109	Not Impacted	Impacted	Not Impacted	Not Impacted
S110	Not Impacted	Impacted	Impacted	Not Impacted
S111	Impacted	Impacted	Not Impacted	Not Impacted
S112	Not Impacted	Impacted	Not Impacted	Impacted

Note:

1. NCO = Non-Chironomid/Oligochaete

Table 9-7. **Onondaga Lake Benthic Community Analysis, 2000 Data**

Station	Water Depth (m)	Total # of Taxa	Taxa Richness (avg. of 5 reps.)	Dominance Index	Total # of NCO	NCO (avg. of 5 reps.)	Community Composition (PMA)	Shannon- Wiener (Diversity Index)
S305	4	24	16	63%	5	3	28	1.8
S317	3.5	12	8	91%	2	<1	22	1.7
S323	3.5	15	9	86%	2	1	23	1.9
S332	4	19	10	88%	4	2	28	1.6
S337	5	15	9	83%	3	2	22	1.9
S342*	4	22	11	87%	10	4	42	2
S344	3.5	16	8	90%	6	2	33	1.8
S365*	4	26	14	71%	4	3	47	2.5
S372*	1.5	36	22	84%	6	4	54	3.1
OT-6*	5	33	19	77%	15	9	60	2.4

Notes:

1. Stations deeper than 5 m not included (see text).
2. NCO = Non-Chironomid/Oligochaete

* Zebra mussels comprised a percentage of the population.

Table 9-8. Onondaga Lake and Otisco Lake Benthic Stations for Evaluation - Impairment Assessment, 2000 Data

Station	Total # of Taxa	Dominance Index	Total # of NCO	Community Composition	Shannon- Wiener (Diversity Index)	Impairment Assessment
S305	24 = M	63% = S	5 = M	28 = I	1.8 =M	Moderately impaired
S317	12 = I	91% = I	2 = I	22 = I	1.7 = M	Severely impaired
S323	15 = M	86% = M	2 = I	23 = I	1.9 = M	Moderately impaired
S332	19 = M	88% = M	4 = I	28 = I	1.6 = M	Moderately impaired
S337	15 = M	83% = M	3 = I	22 = I	1.9 = M	Moderately impaired
S342	22 = M	87% = M	10 = S	42 = M	2.0 = M	Moderately impaired
S344	16 = M	90% = M	6 = M	33 = I	1.8 = M	Moderately impaired
S365	26 = S	71% = S	4 = I	47 = M	2.5 = S	Slightly impaired
S372	36 = N	84% = M	6 = M	54 = S	3.1 = N	Slightly impaired*
OT-6	33 = N	77% =M	15 = S	60 = S	2.4 = S	Slightly impaired

Notes:

1. N = Non-impaired; S = Slightly impaired; M = Moderately impaired; I = Severely impaired
2. Stations deeper than 5 m not included (see text).

* Three or more of the five metrics did not exhibit the same impairment category; professional judgement and the results of all five metrics used for impairment assessment.

Table 9-9. Onondaga Lake and Otisco Lake Benthic Stations Statistical Evaluation, 2000 Data

Station	Total # of Taxa	Dominance Index	Total # of NCO	Shannon- Wiener (Diversity Index)
S305	Impacted	Not Impacted	Impacted	Not Impacted
S317	Impacted	Not Impacted	Impacted	Impacted
S323	Impacted	Not Impacted	Impacted	Impacted
S332	Impacted	Not Impacted	Impacted	Impacted
S337	Impacted	Not Impacted	Impacted	Impacted
S342	Impacted	Not Impacted	Impacted	Not Impacted
S344	Impacted	Not Impacted	Impacted	Impacted
S365	Impacted	Not Impacted	Impacted	Not Impacted
S372	Not Impacted	Not Impacted	Impacted	Not Impacted

Note:
1. NCO = Non-Chironomid/Oligochaete

Table 9-10. Onondaga Lake Tributary Stations for Evaluation - Impairment Assessment, 1992 Data

Station		Total # of Taxa	Dominance Index	Total # of NCO	Community Composition	Shannon- Wiener (Diversity Index)	Impairment Assessment
T1	(Harbor Brook)	25	94%	14	-----	1.2	Severely Impaired
T3	(Onondaga Creek)	11	94%	3	-----	1.5	Severely Impaired
T5	(Ley Creek)	25	95%	11	-----	1	Severely Impaired
T7	(East Flume)	18	80%	3	-----	2	Moderately Impaired
T9	(Tributary 5A)	9	99%	4	-----	1.1	Severely Impaired
T11	(Bloody Brook)	41	58%	20	-----	3.4	Non-impaired
T13	(Ninemile Creek)	20	87%	9	-----	1.9	Moderately Impaired
T15	(Sawmill Creek)	43	53%	27	-----	3.5	Non-impaired

Note:

1. For Community Composition, each tributary station is designated as "-----" - cannot compare tributary stations to a model lake community.

Table 9-11. Site-Specific Sediment Effect Concentrations for Onondaga Lake - 1992 Toxicity Data

		Amphipod Biomass					Amphipod Survival				
Parameter	Units	AET	ER-M	ER-L	TEL	PEL	AET	ER-M	ER-L	TEL	PEL
Metals											
Arsenic	mg/kg-dw	5.4	5.1	3.2	2.2	4.4				2.8	5.2
Cadmium	mg/kg-dw	8.6	2.5	1.1	1.5	3.2		3.0	3.0	2.5	3.5
Chromium	mg/kg-dw	195	107.5	23	31.5	92.2	690	1,990	1,990	301	431
Lead	mg/kg-dw	121	96.4	11	23.3	86.7		238	238	78	147
Mercury	mg/kg-dw	20.4	6	0.6	1.2	4.3		5.2	5.2	3	5.7
Nickel	mg/kg-dw	50	31.1	18.6	14.9	27	215	650	650	101	144
Zinc	mg/kg-dw	218	160	61.1	73.3	159		185	185	123	184
VOCs											
Benzene	µg/kg-dw	5,300	221	13	21	395	5,300	5,700	5,700	434	2,104
Chlorobenzene	µg/kg-dw	10,005	1,100	49	83	961		30,000	30,000	1,817	9,950
Ethylbenzene	µg/kg-dw		657	142	206	657				657	1,107
Toluene	µg/kg-dw	443	100	18	20	99		810	810	133	426
Xylene (Total)	µg/kg-dw	606	2,800	570	679	1,282				606	6,880
SVOCs											
2-Methylnaphthalene	µg/kg-dw	1,063	3,700	3,700	1,201	1,788				620	2,118
Acenaphthene	µg/kg-dw		1,300	782	686	1,144		1,400	1,400	917	1,420
Anthracene	µg/kg-dw		100	42.8	62	248		630	630	237	633
Benzo(a)anthracene	µg/kg-dw		240	58.9	121	457		1,400	1,400	511	1,109
Benzo(a)pyrene	µg/kg-dw		155	56.6	111	343				180	691
Benzo(g,h,i)perylene	µg/kg-dw		550	202	341	1,011		3,000	3,000	1,149	2,510
Benzo(k)fluoranthene	µg/kg-dw		210	67.6	98	323		580	580	285	522
Chrysene	µg/kg-dw		365	82.9	207	640		2,200	2,200	799	1,468
Dibenz(a,h)anthracene	µg/kg-dw		130	58	74	248		530	530	221	463

Table 9-11. (cont.)

		Amphipod Biomass					Amphipod Survival				
Parameter	Units	AET	ER-M	ER-L	TEL	PEL	AET	ER-M	ER-L	TEL	PEL
SVOCs (cont.)											
Dibenzofuran	µg/kg-dw				340	802				340	802
Dichlorobenzenes (Sum)	µg/kg-dw	1,373	2,735	75	82	1,415		1,300	1,300	466	3,695
Fluoranthene	µg/kg-dw		1,800	336	650	3,036		28,000	28,000	5,103	13,299
Fluorene	µg/kg-dw		520	166	128	536		520	520	228	676
Hexachlorobenzene	µg/kg-dw	290	120	11	13	86		16	16	18	46
Indeno(1,2,3-cd)pyrene	µg/kg-dw		150	58	122	283		630	630	327	630
Naphthalene	µg/kg-dw	11,000	1,800	385	579	1,754		2,300	2,300	1,322	3,485
PAH-high MW	µg/kg-dw		37,300	37,300	37,300	37,300				37,300	37,300
PAH-low MW	µg/kg-dw		45,400	45,400	45,400	45,400				45,400	45,400
Phenanthrene	µg/kg-dw		350	96	157	522		2,000	2,000	707	1,646
Phenol	µg/kg-dw	45	45	45	45	45				45	45
Pyrene	µg/kg-dw		460	69	173	842		3,300	3,300	1,149	2,312
Trichlorobenzenes (Sum)	µg/kg-dw	287	1,300	708	183	527				423	2,025
Pesticides and PCBs											
alpha-Chlordane/Chlordane (sum)	µg/kg-dw				5	5				5	5
Aroclor-1016	µg/kg-dw	90	180	180	127	127				135	167
Aroclor-1248	µg/kg-dw	750	320	95	153	382	750	1,100	1,100	503	769
Aroclor-1254	µg/kg-dw				77	93				77	93
Aroclor-1260	µg/kg-dw	900	240	120	147	392	900	1,000	1,000	469	736
DDT and metabolites	µg/kg-dw	16	30	16	14	21				14	33
PCBs (Sum)	µg/kg-dw	1,650	400	141	190	568	1,650	2,100	2,100	744	1,322

Table 9-11. (cont.)

Parameter	Units	Chironimid Biomass AET	Chironimid Biomass ER-M	Chironimid Biomass ER-L	Chironimid Biomass TEL	Chironimid Biomass PEL	Chironimid Survival AET	Chironimid Survival ER-M	Chironimid Survival ER-L	Chironimid Survival TEL	Chironimid Survival PEL
Metals											
Arsenic	mg/kg-dw	5.4	4.9	3.9	2.4	4.2	4.3	4.4	0.9	1.3	3.6
Cadmium	mg/kg-dw		2.8	1.1	1.4	3.4		2.1	0.9	1.4	3.1
Chromium	mg/kg-dw	389	71.2	28.8	36.1	77.9	389	47.9	17.6	29.3	67.3
Lead	mg/kg-dw	167	83.5	31.0	32.5	81.1	116	56.9	9.7	13.3	57.6
Mercury	mg/kg-dw	30	5.2	0.8	1.3	4.1	13	2.8	0.5	1.0	2.8
Nickel	mg/kg-dw	76.4	25.4	16.2	14.1	27.8	72.1	20.9	5.2	8.4	25.8
Zinc	mg/kg-dw		153	64.9	76.6	169.4	270	94.6	37.9	56.7	120.3
VOCs											
Benzene	µg/kg-dw	5,300	400.5	28.5	30.4	535.8	5,300	42.0	27.3	42.4	299
Chlorobenzene	µg/kg-dw	10,005	840	25.8	67.8	1,717	10,005	580	64.4	48.3	799
Ethylbenzene	µg/kg-dw	13.3	1,300	1,300	131.2	131.2		657	142	206	657
Toluene	µg/kg-dw		29.0	16.7	19.3	78.2	443	27.5	13.1	15.9	50.3
Xylene (Total)	µg/kg-dw	2,800	13,000	13,000	2,656	4,854	606	1,640	153	367	997
SVOCs											
2-Methylnaphthalene	µg/kg-dw	1,063	3,700	3,700	1,201	1,788	1,063	1,930	514	655	1,340
Acenaphthene	µg/kg-dw		1,300	750	635	1,193	1,700	1,200	469	478	1,033
Anthracene	µg/kg-dw		210	61	71	359	4,400	210	33	50	249
Benzo(a)anthracene	µg/kg-dw		420	177	191	600		415	61	118	451
Benzo(a)pyrene	µg/kg-dw		255	121	143	440		210	63	98	355
Benzo(g,h,i)perylene	µg/kg-dw		1,600	480	424	1,697	2,700	1,300	228	307	1,172
Benzo(k)fluoranthene	µg/kg-dw		250	121	119	310	1,100	240	63	81	253
Chrysene	µg/kg-dw		515	252	256	707		440	100	172	541
Dibenz(a,h)anthracene	µg/kg-dw		200	82	90	283	730	180	49	68	218

Table 9-11. (cont.)

Parameter	Units	Chironimid Biomass					Chironimid Survival				
		AET	ER-M	ER-L	TEL	PEL	AET	ER-M	ER-L	TEL	PEL
SVOCs (cont.)											
Dibenzofuran	µg/kg-dw				340	802		340	340	295	561
Dichlorobenzenes (Sum)	µg/kg-dw	18,400	1,300	155	129	1,307	1,373	773	22	44	765
Fluoranthene	µg/kg-dw		1,800	648	843	3,338	26,000	1,400	140	483	2,482
Fluorene	µg/kg-dw		375	118	110	476	3,500	305	55	67	327
Hexachlorobenzene	µg/kg-dw	170	47	15	13	75	28	28	7	9	24
Indeno(1,2,3-cd)pyrene	µg/kg-dw		360	111	137	509		370	59	102	503
Naphthalene	µg/kg-dw		1,450	494	599	1,826	2,100	1,400	340	471	1,380
PAH-high MW	µg/kg-dw		37,300	37,300	37,300	37,300		37,300	37,300	37,300	37,300
PAH-low MW	µg/kg-dw		45,400	45,400	45,400	45,400		45,400	45,400	45,400	45,400
Phenanthrene	µg/kg-dw		510	176	199	662	16,000	480	92	135	491
Phenol	µg/kg-dw				45	45	45	45	45	45	45
Pyrene	µg/kg-dw		805	201	347	1,069		650	114	238	795
Trichlorobenzenes (Sum)	µg/kg-dw		663	153	303	1,303	287	930	186	209	482
Pesticides and PCBs											
alpha-Chlordane/Chlordane (sum)	µg/kg-dw				5	5				5	5
Aroclor-1016	µg/kg-dw	90	180	180	127	127		135	99	104	135
Aroclor-1248	µg/kg-dw	670	410	182	194	417	470	300	82	99	307
Aroclor-1254	µg/kg-dw		77	77	80	85	77	83	69	74	80
Aroclor-1260	µg/kg-dw	380	260	113	155	267	240	240	80	115	221
DDT and metabolites	µg/kg-dw	16	47	47	24	27	16	47	47	24	27
PCBs (Sum)	µg/kg-dw	1,050	800	142	201	660	710	400	136	151	382

Notes:

[a] All concentrations in dry weight

[b]Maps of exceedances of ER-L, ER-M, TEL, and PEL values are presented in Appendix F.

1. Effects values could not be calculated for some parameters and are shown as blanks. See text for discussion.

AET - apparent effects threshold
BTX - benzene, toluene, xylenes
ER-L - effects-range low
ER-M - effects-range median
PCB - polychlorinated biphenyl
PAH - polycyclic aromatic hydrocarbon
PEL - probable effect level
TEL - threshold effect level

Table 9-12. Site-Specific Sediment Effect Concentrations for Onondaga Lake - 2000 Toxicity Data

		Amphipod Survival					Amphipod Biomass					Amphipod Reproduction				
Parameter	Units	AET	ER-M	ER-L	TEL	PEL	AET	ER-M	ER-L	TEL	PEL	AET	ER-M	ER-L	TEL	PEL
Metals																
Arsenic	mg/kg-dw	6	8.7	6.0	5.7	6.9		4.0	3.7	4.6	5.9	8.9	9.1	8.0	6.2	7.9
Cadmium	mg/kg-dw		2.3	1.2	2.0	3.1	4.2	7.8	4.1	3.5	5.4		1.4	1.1	1.9	2.4
Chromium	mg/kg-dw	148	114.3	51	69.2	111		115	89	87	136		66.2	41.9	61.1	107
Lead	mg/kg-dw	116	110.8	46.2	73.3	108		117	96	89	122	172	71.3	31.1	57.5	94.4
Mercury	mg/kg-dw	9.6	9.2	0.7	1.4	7.1		9.4	3.2	3.4	10.1	17.2	0.7	0.7	1.4	2.8
Nickel	mg/kg-dw	40.6	53.5	27.8	30.4	43.9		35.5	34.5	32.4	46.1		49.8	27.0	30.7	50.4
Zinc	mg/kg-dw	323	126.5	77	144.5	187		296	274	190	278		96.3	65.4	130	169
VOCs																
Benzene	µg/kg-dw	73	275	64	36	116		14	14	32	72		180	180	60	259
Chlorobenzene	µg/kg-dw	140	2,800	282	123	499		208	86	137	3,584	65,000	60,115	12,207	1,332	12,688
Ethylbenzene	µg/kg-dw	12	1,600	792	72	125				12	1,650	1,600	1,345	741	76	933
Toluene	µg/kg-dw	90	233	51	46	128		5	5	16	35	90	460	460	67	172
Xylene (Total)	µg/kg-dw	6	7,800	1,696	121	216				170	8,640	170	8,850	8,010	221	1,034
SVOCs																
Acenaphthene	µg/kg-dw		580	244	286	580		160	160	400	400				580	874
Anthracene	µg/kg-dw	270	660	308	272	408	270	1,100	1,100	486	522				220	602
Benzo(a)anthracene	µg/kg-dw	680	700	380	383	652	700	1,075	415	429	852				370	693
Benzo(a)pyrene	µg/kg-dw	810	1,215	507	518	912	810	1,175	435	487	896				450	782
Benzo(g,h,i)perylene	µg/kg-dw	700	1,100	1,100	620	782	700	665	317	373	621				385	692
Benzo(k)fluoranthene	µg/kg-dw	660	930	394	445	762	660	920	376	437	758				430	652
Chrysene	µg/kg-dw	940	750	510	493	794	940	1,270	526	559	1,024				505	902
Dibenz(a,h)anthracene	µg/kg-dw	270	430	430	308	331	270	430	430	308	331				245	358
Dibenzofuran	µg/kg-dw		1,400	1,400	1,400	1,400				1,400	1,400				1,400	1,400
Dichlorobenzenes (Sum)	µg/kg-dw	290	1,164	556	326	544		1,164	1,164	699	5,849	6,100	119,880	24,584	3,599	16,955
Fluoranthene	µg/kg-dw	1,400	1,800	968	822	1,536	1,800	2,240	832	895	1,768				795	1,660
Fluorene	µg/kg-dw		930	314	391	930		160	160	522	522				930	1,469
Hexachlorobenzene	µg/kg-dw	54	27	2	5	22		15	6	10	25		14.8	5.4	9.8	24.4
Indeno(1,2,3-cd)pyrene	µg/kg-dw	580	600	280	315	557	580	600	280	315	557				310	566
Naphthalene	µg/kg-dw	310	35,000	9,733	1,866	3,180				16,155	71,000	38,000	32,000	6,552	1,737	29,226
Phenanthrene	µg/kg-dw	550	1,100	582	437	739		660	308	389	1,138	2,500	2,880	1,344	733	1,617
Phenol	µg/kg-dw		1,900	1,900	1,900	1,900				1,900	1,900		1,900	1,900	1,900	1,900
Pyrene	µg/kg-dw	1,300	500	180	372	689	1,300	1,525	665	661	1,173		100	100	251	350
Trichlorobenzenes (Sum)	µg/kg-dw		35,000	35,000	35,000	35,000				35,000	35,000		35,000	35,000	35,000	35,000
Pesticides and PCBs																
Aroclor-1242	µg/kg-dw	1,210	181	164	141	315		689	272	229	455	1,210	5,331	1,195	536	1,266
Aroclor-1254	µg/kg-dw	445	98	34	52	129		272	133	98	216	445	345	108	101	244
Aroclor-1260	µg/kg-dw	129	56	12	21	66		82	45	41	84	129	93	29	35	85
Chlordane (Sum)	µg/kg-dw		1.5	1.2	1.8	3.3	6.2	5.6	2.3	2.4	4.7				2.1	6.1
DDT and metabolites (Sum)	µg/kg-dw		4.1	1.3	2.5	6.4	11.4	12.9	9.0	6.1	9.5		3.5	1.7	3.0	5.6
Hexachlorocyclohexanes (Sum)	µg/kg-dw	2.9	5.3	2.6	2.6	3.8		2.1	2.1	2.8	3.7	6.6	5.3	4.2	3.3	5.1
PCBs (Sum)	µg/kg-dw	1,784	265	158	177	388		1,043	450	316	709	1,784	174	148	186	296

Table 9-12. (cont.)

Parameter	Units	Chironimid Survival AET	Chironimid Survival ER-M	Chironimid Survival ER-L	Chironimid Survival TEL	Chironimid Survival PEL	Chironimid Biomass AET	Chironimid Biomass ER-M	Chironimid Biomass ER-L	Chironimid Biomass TEL	Chironimid Biomass PEL	Chironimid Emergence AET	Chironimid Emergence ER-M	Chironimid Emergence ER-L	Chironimid Emergence TEL	Chironimid Emergence PEL
Metals																
Arsenic	mg/kg-dw	6	7.7	3.5	4.2	6.6	6.0	8.1	2.8	4.2	6.7	9.1	8.4	6.4	5.6	6.9
Cadmium	mg/kg-dw	3.7	1.9	0.7	1.6	2.5	4.2	3.0	1.3	2.0	3.2		1.9	1.2	1.9	2.8
Chromium	mg/kg-dw	113	85.5	31.1	52.3	88.2	113	114.3	57.1	73.2	100	148	113	48	66.6	118.8
Lead	mg/kg-dw	142	79.6	36.0	62.2	96.6	116	85.5	18.5	46.5	96.8	142.0	92.0	44.4	67.6	102.2
Mercury	mg/kg-dw	17.2	3.0	0.7	1.6	5.9	9.6	2.3	0.7	1.5	3.9	17.2	3.3	1.6	2.5	5.3
Nickel	mg/kg-dw	40.6	36.8	20.4	26.0	36.3	40.6	43.3	17.2	25.6	38.2	57.1	49.8	29.0	31.5	42.3
Zinc	mg/kg-dw	269	130	89	157	184	282	127	76	143	181	323	130	84	140	190
VOCs																
Benzene	µg/kg-dw	73	275	57	38	116	73	275	63.8	36.4	116.3	20	275	105	38	69
Chlorobenzene	µg/kg-dw	360	1,515	50	70	629	140	1,580	142.5	93.6	395.6	360	2,800	121	93	799
Ethylbenzene	µg/kg-dw	12	1,095	180	43	107	12	1,600	792	71.7	124.9	12	1,095	180	43	107
Toluene	µg/kg-dw	90	232	49	45	128	90	233	50.7	45.7	128.2	90	254	88	29	117
Xylene (Total)	µg/kg-dw	6	7,800	1,696	121	216	6	7,800	1,696	121.5	216.3	6	3,985	55	22	155
SVOCs																
Acenaphthene	µg/kg-dw	160	1,000	1,000	400	400		580	244	286	580	160	1,000	1,000	400	400
Anthracene	µg/kg-dw		180	148	203	391	270	660	308	272	408		215	211	239	428
Benzo(a)anthracene	µg/kg-dw		275	180	349	610	680	500	265	318	557		390	318	338	559
Benzo(a)pyrene	µg/kg-dw		250	226	416	616	810	330	266	351	482		390	342	396	605
Benzo(g,h,i)perylene	µg/kg-dw		240	202	359	470	700	665	317	373	621		420	420	383	557
Benzo(k)fluoranthene	µg/kg-dw		250	210	375	542	660	260	244	325	405		345	277	350	510
Chrysene	µg/kg-dw		395	280	507	803	940	600	373	444	717		560	472	466	773
Dibenz(a,h)anthracene	µg/kg-dw		130	130	187	223	270	430	430	308	331				245	358
Dibenzofuran	µg/kg-dw		1,400	1,400	1,400	1,400		1,400	1,400	1,400	1,400		1,400	1,400	1,400	1,400
Dichlorobenzenes (Sum)	µg/kg-dw	1,164	3,430	522	363	1,626	290	1,164	556	326	544	1,164	760	270	305	828
Fluoranthene	µg/kg-dw		555	415	773	1,253	1,400	1,280	564	665	1,302		830	774	701	1,174
Fluorene	µg/kg-dw	160	1,700	1,700	522	522		930	314	391	930	160	1,700	1,700	522	522
Hexachlorobenzene	µg/kg-dw	53.5	9.0	2.3	6.2	18.3	53.5	15.0	2.5	6.4	18.2	53.5	17.0	3.6	8.0	20.7
Indeno(1,2,3-cd)pyrene	µg/kg-dw		200	188	323	403	580	200	200	257	325		275	215	263	414
Naphthalene	µg/kg-dw	310	35,000	9,733	1,866	3,180	310	35,000	9,733	1,866	3,180	190	35,000	9,817	1,619	2,558
Phenanthrene	µg/kg-dw	1,100	300	174	308	506	550	1,030	275	311	721	1,100	960	346	309	762
Phenol	µg/kg-dw		1,900	1,900	1,900	1,900		1,900	1,900	1,900	1,900		1,900	1,900	1,900	1,900
Pyrene	µg/kg-dw		430	220	556	931	1,300	475	205	420	692		620	524	538	891
Trichlorobenzenes (Sum)	µg/kg-dw		35,000	35,000	35,000	35,000		35,000	35,000	35,000	35,000		35,000	35,000	35,000	35,000
Pesticides and PCBs																
Aroclor-1242	µg/kg-dw	168	192	103	133	173	328	187	165	129	201	1,210	181	150	143	315
Aroclor-1254	µg/kg-dw	108	110	25	43	106	173	135	36	52	124	445	140	49	63	149
Aroclor-1260	µg/kg-dw	39.6	75.2	12.5	23.8	53.0	80.9	75.2	12.5	20.5	57.3	129	39.6	12.4	20.5	59.9
Chlordane (Sum)	µg/kg-dw	2.6	3.2	1.4	1.5	2.7	6.2	2.3	1.2	1.7	3.3		2.6	1.4	1.6	4.4
DDT and metabolites (Sum)	µg/kg-dw	8.1	4.1	1.3	2.6	5.0	10.2	4.1	1.3	2.5	5.2		4.3	1.3	2.6	6.5
Hexachlorocyclohexanes (Sum)	µg/kg-dw	2.1	5.3	3.2	2.6	3.3	2.9	5.3	2.6	2.6	3.8	2.9	6.6	4.4	3.1	4.2
PCBs (Sum)	µg/kg-dw	302	229	72	136	260	582	302	161	169	310	1,784	290	196	187	390

Notes:

[a] All concentrations in dry weight

[b] Maps of exceedances of ER-L, ER-M, TEL, and PEL values are presented in Appendix F.

1. Effects values could not be calculated for some parameters and are shown as blanks. See text for discussion.

AET - apparent effects threshold
BTX - benzene, toluene, xylenes
ER-L - effects-range low
ER-M - effects-range median
PCB - polychlorinated biphenyl
PAH - polycyclic aromatic hydrocarbon
PEL - probable effect level
TEL - threshold effect level

Table 9-13. **Comparison of Various Site-Specific Sediment Effect Concentrations and Probable Effect Concentrations for Onondaga Lake, 1992 Data[a,b]**

	AET	ER-L	ER-M	TEL	PEL	PEC
Metals (mg/kg)						
Antimony	NC	3.1	3.1	4	4.3	3.6
Arsenic	4.3	0.90	4.4	1.29	3.55	2.4
Cadmium	8.6	0.94	2.1	1.42	3.11	2.4
Chromium	195	17.6	47.9	29.3	67.3	50.3
Copper	83.7	12.3	40.7	19.1	48.3	32.9
Lead	116	9.68	56.9	13.3	57.6	34.5
Manganese	445	197	280	231	295	278
Total mercury	13	0.51	2.8	0.99	2.84	2.2
Nickel	50	5.22	20.9	8.37	25.8	16.4
Selenium	0.94	0.42	0.6	0.4	0.68	0.58
Silver	2.7	0.82	1.2	0.9	1.42	1.28
Vanadium	12.2	2.7	6	3.4	8.3	5.6
Zinc	218	37.9	94.6	56.7	12	88
Organic Compounds						
BTEX Compounds (µg/kg)						
Benzene	5,300	27.3	42	42.4	299	150
Ethylbenzene	13.3	142	657	206.0	657	176
Toluene	443	13.1	27.5	15.9	50.3	41.8
Xylenes	606	153	1,640	367	997	560.8
Chlorinated Benzenes (µg/kg)						
Chlorobenzene	10,000	64.4	580	48.3	799	428
Dichlorobenzenes	1,373	21.5	773	44.2	765	239
Trichlorobenzenes	287	186	930	209	482	347
Hexachlorobenzene	28	7.16	28	8.9	23.6	16.4
Polychlorinated Biphenyls (µg/kg)						
Aroclor 1016	90	99	135	104	135	111
Aroclor 1248	470	82	300	99	307	204
Aroclor 1254	77	68.5	82.5	74	79.7	76
Aroclor 1260	240	80	240	115	221	164
Total PCBs	710	136	400	151	382	295
PAH Compounds (µg/kg)						
Naphthalene	2,100	340	1,400	471	1,380	917
Acenaphthene	1,700	469	1,200	478	1,030	861
Fluorene	3,500	55.2	305	66.9	327	264
Phenanthrene	16,000	92.2	480	135	491	543
Anthracene	4,400	33	210	49.6	249	207
Fluoranthene	26,000	140	1,400	483	2,482	1,436
Pyrene	NC	114	650	238	795	344
Benz[a]anthracene	NC	60.7	415	118	451	192
Chrysene	NC	100	440	172	541	253
Benzo[b]fluoranthene	1,100	63.1	240	80.9	253	908
Benzo[a]pyrene	NC	62.8	210	98.2	355	146
Indeno[1,2,3-cd]pyrene	NC	58.8	370	102	503	183
Dibenz[a,h]anthracene	730	49.4	180	67.7	218	157
Benzo[ghi]perylene	2,700	228	1,300	307	1,170	780
Acenapthylene	3,000	507	1,850	673	1,970	1,301
Benzo[k]fluoranthene	1,100	63.1	240	80.9	253	203
Dibenzofuran	NC	340	340	295	561	372

Table 9-13. (cont.)

	AET	ER-L	ER-M	TEL	PEL	PEC
Other SVOCs (µg/kg)						
Phenol	45	45	45	45	45	45
Pesticides (µg/kg)						
DDT and Metobolites	16.3	47	47	23.7	26.6	29.6
Chlordane	NC	NC	NC	5.08	5.08	5.1
Heptachlor and Heptachlor Epoxide	NC	NC	NC	NC	NC	NC
Dioxins/Furans						
Total Dioxins/Furans	NC	NC	NC	NC	NC	NC

Notes:

[a] All concentrations in dry weight

[b]Maps of exceedances of ER-L, ER-M, TEL, PEL and PEC values are presented in Appendix F.

AET - apparent effects threshold
BTX - benzene, toluene, xylenes
ER-L - effects-range low
ER-M - effects-range median
NC - value was not calculated because of an insufficient number of detected observations or data points
PCB - polychlorinated biphenyl
PAH - polycyclic aromatic hydrocarbon
PEL - probable effect level
TEL - threshold effect level
PEC - Probable Effect Concentration

Table 9-14. Comparison of Lowest AET Values for Toxicity Endpoints Found for Onondaga Lake in 1992 and 2000

	AET (mg/kg dry weight)		AET
Chemical	1992	2000	Ratios[a]
Arsenic	4.3	6	1.4
Cadmium	8.6	4.2	2.1
Chromium	195	113	1.7
Lead	116	116	1.0
Mercury	13	9.4	1.4
Nickel	50	40.6	1.2
Zinc	218	269	1.2

Notes:

AET - apparent effects threshold

[a] The ratio for each chemical was calculated as the higher AET (i.e., regardless of year) divided by the lower AET.

Table 9-15. Toxicity Reference Values for Fish

Chemical of Concern	NOAEL/LOAEL	Reference
Antimony	5.0/9.0 mg/kg ww tissue	Doe et al. (1987)
Arsenic	0.5/1.3 mg/kg ww tissue	NRCC (1978)/Walsh (1977)
Chromium	0.23/0.78 mg/kg ww tissue	Van Der Putte et al. (1981a)
Mercury/Methylmercury	0.1/ 0.3 mg/kg ww tissue	NOAA (2002)
Selenium	0.11/1.1 mg/kg ww tissue	Lemly (1997) and references cited therein
Vanadium	0.041/0.41mg/kg ww tissue	Hilton and Bettger (1988)
Zinc	34/40 mg/kg ww tissue	Spehar (1976)
Endrin	0.024/0.24 mg/kg ww tissue	Jarvinen and Tyo (1978)
DDT and metabolites	0.6/2.9 mg/kg ww tissue	Macek (1968)
Polychlorinated biphenyls	1.9/9.3 mg/kg ww tissue	Hansen et al. (1974)
Dioxins/furans	0.29/0.6 µg TEQs/kg lipid	Walker et al. (1994)

Notes:

NOAEL/LOAEL - no-observed-adverse-effect level/lowest-observed-adverse-effect level.

Table 9-16. **Toxicity Reference Values for Avian Receptors**

Chemical of Concern	NOAEL/LOAEL mg/kg-day	Exposure Period	Reference
Inorganic Analytes			
Arsenic	2.46/7.38	7 months	USFWS (1969)
Barium	20.8/41.7	4 weeks	Johnson et al. (1960)
Cadmium	1.45/20	90 days	White and Finley (1978)
Chromium	1.0/5.0	303 days	Haseltine et al. (unpublished)
Copper	47/61.7	10 weeks	Mehring et al. (1960)
Lead	1.18/11.8	12 weeks	Edens et al. (1976)
Mercury (inorganic)	0.45/0.90	up to 365 days	Hill and Schaffner (1976)
Nickel	77.4/107	90 days	Cain and Pafford (1981)
Selenium	0.4/0.8	100 days	Heinz et al. (1989)
Thallium	NA		
Vanadium	11.4/114	12 weeks	White and Dieter (1978)
Zinc	14.5/131	44 weeks	Stahl et al. (1990)
Organic Compounds			
Bis(2-ethylhexyl)phthalate	1.1/11	4 weeks	Peakall (1974)
DDT and metabolites	0.0028/0.028	>365 days	Anderson et al. (1975)
Dichlorobenzenes	6/60	35 days	Hollingsworth et al. (1956)
Endrin	0.01/0.1	>83 days	Fleming et al. (1982)
Hexachlorocyclohexanes	0.11/0.34	7 days	Jansen (1976)
Methylmercury	0.0064/0.064	3 generations of mallard ducks	Heinz (1974, 1976a,b, 1979)
Polychlorinated biphenyls (PCBs)	0.18/1.8	16 weeks	Dahlgren et al. (1972)
Polycyclic aromatic hydrocarbons	0.143/1.43	151 days	Hough et al. (1993)
Trichlorobenzenes	NA	NA	NA
Xylenes	NA	NA	NA
Dioxins/furans	0.000014/ 0.00014	70 days	Nosek et al. (1992)

Notes:

NOAEL/LOAEL - no-observed-adverse-effect level/lowest-observed-adverse-effect level.
NA - No appropriate avian study available.
Units are mg/kg-day (dietary dose for wildlife TRVs).

Table 9-17. Toxicity Reference Values for Mammalian Receptors

Chemical of Concern		NOAEL/LOAEL mg/kg-day	Exposure Period	Reference
Inorganic Analytes				
Antimony		0.125/1.25	mouse lifetime (> 1 year)	Schroeder et al. (1968)
Arsenic		0.126/1.26	3 generations of mice	Schroeder and Mitchener (1971)
Barium		45/75	105 weeks	NTP (1994)
Cadmium		1/10	42 days	Sutou et al. (1980)
Chromium		3.28/13.14	365 days/90 days	Mackenzie et al. (1958)/ Steven et al. (1976)
Copper		11.7/15.14	357 days	Aulerich et al. (1982)
Lead		8/80	3 generations of rats	Azar et al. (1974)
Manganese		88/284	224 days	Laskey et al. (1982)
Mercury (inorganic)		1.0/10	155 days	Aulerich et al. (1974)
Nickel		40/80	3 generations of rat	Ambrose et al. (1976)
Selenium		0.20/0.33	2 generations of rat	Rosenfeld and Beath (1954)
Thallium		0.74/0.074	60 days	Formigli et al. (1986)
Vanadium		0.21/2.1	180 days	Domingo et al. (1986)
Zinc		160/320	16 days of gestation	Schlicker and Cox (1968)
Organic Compounds				
Chlordane		0.15/0.75	104 weeks	Khasawinah and Grutsch (1989)
DDT and metabolites		0.8/4	2 years	Fitzhugh (1948)
Dieldrin		0.009 /0.018	up to 336 days	Harr et al. (1970)
Methylmercury		0.0025/0.025	2 years	Wobeser et al. (1976) and Wren et al. (1987)
Hexachlorobenzene		0.014/0.14	331 days	Bleavins et al. (1984)
PCBs	mink, otter	0.0034/0.034	2 generations of mink	Restum et al. (1998)
	bat, shrew	0.4/1.6	2 generations of rats	Linder et al. (1974)
Polycyclic aromatic hydrocarbon		1/10	9 days	Mackenzie and Angevine (1981)
Trichlorobenzenes		14.8/53.6	3 generations of rat	Robinson et al. (1981)
Xylenes		2.1/2.6	days 6 to 15 of gestation	Marks et al. 1982
Dioxins/furans		0.000001/0.00001	3 generations of rat	Murray et al. (1979)

Notes:

NOAEL/LOAEL - no-observed-adverse-effect level/lowest-observed-adverse-effect level.

Units are mg/kg-day (dietary dose for wildlife TRVs).

10. RISK CHARACTERIZATION (ERAGS STEP 7)

Risk characterization evaluates the likelihood of adverse effects occurring as a result of exposure to chemicals and/or stressors, and discusses the qualitative and quantitative assessment of risks to ecological receptors. Risk estimation integrates effects information (Chapter 9) with exposure profiles (Chapter 8) to provide an estimate of risk (this chapter) and related uncertainties (Chapter 11). Assessment endpoints and the associated measurement endpoints selected during problem formulation (Chapter 6) are evaluated to describe potential risks to receptors, as detailed below.

10.1 Assessment Endpoint: Sustainability of a Macrophyte Community That Can Serve as a Shelter and Food Source for Local Invertebrates, Fish, and Wildlife

10.1.1 Does the Macrophyte Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern?

Measurement Endpoint: Comparison of the Onondaga Lake Macrophyte Community to Reference Lakes

As described in Chapter 9, Section 9.1, the macrophyte community of Onondaga Lake has exhibited low diversity and abundance since at least 1940. A typical lake in New York State has an average of 18 species of aquatic plants: 14 submerged, two floating-leaved, and two emergent species (Madsen et al., 1993), and the average eutrophic lake in New York State has 15 species of macrophytes (Madsen et al., 1993). Ten species of macrophytes are currently found in Onondaga Lake (Madsen et al., 1998). High salinity, low visibility, eutrophication, and the poor substrata of Onondaga Lake are likely to have been factors in the decline of species richness.

The density of macrophytes in Onondaga Lake is also quite low. Only 13 percent of 3,498 quadrants surveyed in 1991 had aquatic plants. Sago pondweed (*Potamogeton pectinatus*), a species able to tolerate high salinities, was the dominant, and often the only, species observed (Madsen et al., 1993). Onondaga Lake shows both reduced macrophyte abundance and diversity as compared to reference lakes (Madsen et al., 1993). These results indicate that macrophytes have been extirpated from many areas of Onondaga Lake. Even with remediation and revegetation efforts it could take years before the aquatic macrophytes fully recover.

10.1.2 Do the Chemicals/Stressors Present in Onondaga Lake Affect Macrophyte Growth and Survival?

Measurement Endpoint: Greenhouse Growth Studies and Macrophyte Transplant Studies

In a series of greenhouse studies described in Chapter 9, Section 9.1.1, Madsen et al. (1993, 1996) found that growth on the fertile reference sediment was significantly higher than growth on Onondaga Lake sediments. They predicted that improvement in water clarity or quality alone would not improve plant growth, as sediment degradation is directly related to the input of calcium chloride ($CaCl_2$) into the lake and the resulting calcium carbonate deposition. Even plant leaves and stems were coated with calcium carbonate particles (Madsen et al., 1993).

The macrophyte transplant conducted in 1992 showed macrophyte survival to be minimal at Onondaga Lake, in contrast to higher survival rates seen at the reference lake (Otisco Lake). Habitat restoration efforts are underway, but the limiting factors present in the lake make it difficult to introduce new species and for the macrophytes currently in the lake to spread.

The Onondaga Lake macrophyte community is considered to be adversely affected by the ionic waste discharged into Onondaga Lake. This waste has increased salinity concentrations, decreased water transparency, degraded lake sediments, and created conditions for oncolite formation. The combination of wave action, sediment characteristics, and sparse vegetation results in low plant colonization rates (Madsen et al., 1998).

10.1.3 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Values and Qualitative Evaluation of Narrative Standards

There are no standards that specifically address risk to macrophytes, and therefore the potential risk to macrophytes due to exceedances of water quality standards is unknown. New York State has narrative water quality standards (6 NYCRR Part 703.2), which regulate physical parameters and aesthetic conditions that impair the best use of the surface water but may not be physically measurable.

The very high concentrations of calcite ($CaCO_3$) in the lake result in contravention of several of these narrative standards including the prohibitions for suspended, colloidal or settleable solids, and turbidity. When calcite becomes resuspended into the water column during normal wave action, the standard for turbidity is exceeded. Calcite coming out of solution slowly settles to the bottom. The high concentrations of calcite that have been deposited onto the surfaces of macrophytes in Onondaga Lake in the past may have been sufficient to completely coat plants (Auer et al., 1996a). This mechanism may have also been responsible for, or contributed to, the disappearance of charophytes from Onondaga Lake (Dean and Eggleston, 1984). The decrease in macrophytes caused by calcite deposition and formation of oncolites has impaired the best use of the water (Chapter 3, Section 3.2.4.1) through impairment of the fish population, as described in more detail in Section 10.6.2.

High concentrations of nutrients may also influence macrophyte growth. The high concentrations of ammonia, nitrite, phosphorus, and sulfide in Onondaga Lake are in part a result of total loads received from the lake from the Metropolitan Syracuse Sewage Treatment Plant (Metro) (e.g., Matthews et al., 2000). Post-1992 sampling continues to reflect a eutrophic lake, but conditions appear to have improved due to upgrades to the Metro facility and closure of the chlor-alkali facility (which has led to less alteration of the density stratification/mixing regime). Currently, upgrades to Metro are being guided by an Amended Consent Judgment (ACJ) from 1998 and decreases in effluent concentrations have been made in the last several years (e.g., Matthews et al., 2001). Under the ACJ, Onondaga County is to reduce stressors in Metro effluent over two intervals by December 2012.

The high salinity of Onondaga Lake may also preclude some macrophyte species. Salinity has dropped from 3.3 parts per thousand (ppt) in 1981 to 1.1 ppt (Effler et al., 1996; Onondaga Lake Partnership [OLP], 2002), but is still over an order-of-magnitude greater than the average world river salinity (0.1 ppt).

Low dissolved oxygen (DO) can be a factor in limiting macrophyte growth. However, the Onondaga Lake littoral zone (where macrophytes are found) is considered to extend out into the lake 100 m, or to a depth of 3 m (Madsen et al., 1993). Levels of DO remain at acceptable levels at these depths, with the exception of the week of fall turnover. Even during this period, the lowest recorded DO concentration was 3.4 mg/L. Therefore, DO is not considered a major limiting factor to macrophyte growth.

Visibility may also limit macrophyte growth. In 1992, Secchi depths were generally less than 2 m throughout the year and increased to up to almost 6 m after the fall turnover (Chapter 8, Figure 8-21). The 1997 to 2001 data indicate improvement in visibility, with increased visibility in May and June (Appendix I, Table I-20).

10.2 Assessment Endpoint: Sustainability of a Phytoplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

10.2.1 Does the Phytoplankton Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern?

Measurement Endpoint: Field Observations of the Onondaga Lake Phytoplankton Community

In general, the characteristics of the phytoplankton communities of Onondaga Lake have reflected the polluted and eutrophic nature of the lake. Concentrations of nutrients have also influenced both the types of species found in the lake and the densities of those species (Auer et al., 1996a).

Contaminants present in Onondaga Lake may also affect the phytoplankton community secondarily by influencing the lake community of species feeding on phytoplankton, such as zooplankton (see Section 10.3), which then affect phytoplankton abundance and diversity. This in turn may impact the number and species of higher trophic levels, such as fish.

Although the effect of mercury contamination on the phytoplankton community is unknown, it is evident from the bioaccumulation investigation (PTI, 1993b) that mercury accumulates in phytoplankton and can be passed on to animals feeding on phytoplankton in Onondaga Lake.

10.2.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Values and Qualitative Evaluation of Narrative Standards

There are no standards that specifically address risk to phytoplankton, and therefore the potential risk to phytoplankton due to exceedances of water quality standards is unknown. The summed concentration of total ammonia and nitrate has continuously exceeded levels associated with limitation of phytoplankton growth (Chapter 8, Figure 8-10). Concentrations of nitrate increase in the epilimnion during the summer and early fall and decrease in the hypolimnion during this period. Narrative water quality standards (6 NYCRR Part 703.2) have been exceeded in the lake, specifically those for settleable solids (e.g., calcite), which may physically impact phytoplankton.

10.3 Assessment Endpoint: Sustainability of a Zooplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

10.3.1 Does the Zooplankton Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern?

Measurement Endpoint: Studies of Historical Changes of the Onondaga Lake Zooplankton Community and Associated Contaminant/Stressors

The composition of zooplankton communities in Onondaga Lake has been affected by stressors, including salinity and calcium carbonate deposition. Chloride/salinity levels of Onondaga Lake before the closure of the Honeywell facility were near the upper limit for freshwater organisms, which affected the osmoregulation capabilities of resident zooplankton. As a result of the high salinity and pollution, native species of Daphnia were replaced by exotic high-salinity-tolerant species such as *Daphnia exilis* and *D. curvirostris* during the peak industrial pollution period from the 1950s to the 1980s (Hairston et al., 1999; Duffy et al., 2000). Calcium carbonate particles may have also influenced zooplankton community structure by physically interfering with zooplankton feeding (Auer et al., 1996a).

It has been hypothesized that the successful invasion of exotic Daphnia species was heavily influenced by the absence of effective feeding on zooplankton by fish, as fish species diversity was lower during periods of high salinity (Hairston et al., 1999). As the salinity declined in the 1980s, exotic Daphnia species disappeared and were replaced by native species, such as *D. pulicaria* and *D. ambigua* (Hairston et al., 1999). Despite recent increases in zooplankton diversity, the zooplankton assemblage of the lake remains depauperate compared to other lakes in the region (Auer et al., 1996a). The period of peak mercury concentrations in the sediments (based on ^{210}Pb dating) coincides with zero hatching success of *D. exilis* eggs in laboratory monitoring (Hairston et al., 1999). Whether mercury in the water column caused the eggs to become non-viable at the time they were produced, or mercury and/or other chemicals and stressors in the sediments made the eggs non-viable over the burial period, is uncertain.

10.3.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: **Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance and Qualitative Evaluation of Narrative Standards**

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC and USEPA water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). The frequency of exceedances in Onondaga Lake and tributary water varied by contaminant, year, location, and depth, as summarized in the following paragraphs. The total number of samples analyzed for each COC is provided in Table D-1 for 1992 samples, and Table D-46 for 1999 samples. As discussed in Chapter 8, Section 8.1.1, the 1999 sampling was oriented toward collecting data for the Onondaga Lake Human Health Risk Assessment (HHRA) (TAMS, 2002a), and focused on areas where people may be exposed to lake water.

With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, zinc, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality criteria (Chapter 4, Table 4-4).

There were exceedances of mercury standards in a total of 167 samples, 147 of which were collected in 1992 and 20 of which were collected in 1999. Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC mercury wildlife value, as discussed below, but not the chronic water quality value for the protection of aquatic organisms.

The 147 surface water samples from 1992 that exceeded the NYSDEC mercury wildlife standard were collected from the following locations in the lake, its tributaries, and Metro discharge (see Chapter 7, Figure 7-1):

- 20 from the East Flume.
- 20 from Tributary 5A.
- 19 from Harbor Brook.
- 18 from Ninemile Creek.
- 16 from Onondaga Creek.
- 14 from Metro.
- 12 from Ley Creek.
- 12 from the lake outlet.
- 8 from the southern basin.
- 5 from the northern basin.
- 2 from Bloody Brook.
- 1 from Sawmill Creek.

Of these, four surface water samples analyzed for dissolved mercury exceeded the NYSDEC wildlife standard (three taken from the southern basin and one taken from the northern basin). Dissolved mercury was not measured in the tributaries in 1992.

The 20 surface water samples from 1999 that were measured for total mercury and exceeded the NYSDEC wildlife standard were collected at the following locations in the lake (see Chapter 2, Figure 2-17 of the Remedial Investigation report [TAMS, 2002b] for locations and station designations, as discussed below):

- Five from the southern basin.
- Three from the northern basin.
- Two from Lake Park at Lakeland.
- Two from Lake Park at Galeville.
- One at the Willis Avenue Lakeshore exposure area.
- One at the observed fishing area near Tributary 5A.
- One at the access from fairgrounds.
- One at the beach access near Ninemile Creek.
- One from near the mouth of Harbor Brook.
- One from the park/picnic/playground area south of Sawmill Creek.
- One from the Liverpool boat ramp area.
- One from the lake outlet.

Of these, seven surface water samples analyzed for dissolved mercury exceeded the NYSDEC wildlife standard: two at the access from the fairgrounds and one apiece at the beach access near Ninemile Creek, Lake Park at Lakeland, the park/picnic area/playground area south of Sawmill Creek, the Liverpool boat ramp area, and the lake outlet.

For COCs other than mercury, exceedances were as discussed below.

- **Barium:** All four lake water samples analyzed for barium in 1992 exceeded the USEPA Tier 2 Aquatic Life barium standard. No surface water samples from 1 m or less were taken, so samples taken at depths of 6 and 12 m were used for barium comparisons.

- **Copper:** There were 28 surface water exceedances of the NYSDEC chronic copper standard in 1992. Of these, 11 occurred in Tributary 5A, five apiece were in Ley Creek and Harbor Brook, two apiece were in Metro, Ninemile Creek, and the East Flume, and one was in Bloody Brook. There were also 18 exceedances of the NYSDEC acute standard. Of these, nine occurred in Tributary 5A, four were in Harbor Brook, three were in Ley Creek, and one apiece were in the East Flume and Bloody Brook. No open lake surface water samples exceeded the NYSDEC copper standards in 1999.

- **Lead:** There were 35 surface water exceedances of the NYSDEC chronic lead standard in 1992. Nine of these occurred in Tributary 5A, eight were in Ley Creek, six were in Harbor Brook, four were in Ninemile Creek, three apiece were in the East Flume and Onondaga Creek, and one apiece were in Bloody Brook and the lake outlet. No surface water samples exceeded the NYSDEC lead standards in 1999.

- **Manganese:** There were 12 surface water exceedances of the USEPA Tier 2 Aquatic Life manganese standard, including four in 1992 (one each in the southern basin, Ley Creek, Ninemile Creek, and Tributary 5A) and eight in 1999 (two apiece in the southern and northern basins, and one each in the lake outlet, the access from fairgrounds, Lake Park at Lakeland, and Lake Park at Galeville).

- **Zinc:** There were 22 surface water exceedances of the NYSDEC chronic zinc standard in 1992, only one of which was recorded in the lake (southern basin). The remainder of exceedances were detected in the tributaries, with nine exceedances in the East Flume, six in Tributary 5A, four in Harbor Brook, one in Ley Creek, and one in Bloody Brook. All nine exceedances of the NYSDEC acute zinc standards were in the tributaries, with four in the East Flume, two in Tributary 5A, one in Harbor Brook, one in Ley Creek, and one in Bloody Brook. Zinc was not analyzed in 1999.

- **Chlorobenzene:** There was one surface water exceedance of the NYSDEC chronic chlorobenzene standard in 1992 at the East Flume and one in 1999 at the Willis Avenue Lakeshore area.

- **Dichlorobenzenes:** There were 14 exceedances of the NYSDEC chronic dichlorobenzenes standard in 1992, 12 of which were in the East Flume and two of which were in Harbor Brook. In 1999 there was one exceedance, which occurred at the Willis Avenue Lakeshore area.

- **Trichlorobenzenes:** There was one exceedance of the NYSDEC chronic trichlorobenzenes standard in the southern basin in 1992. Trichlorobenzenes were not analyzed in 1999.

- **Bis(2-ethylhexyl)phthalate:** One of the four lake water samples analyzed for bis(2-ethylhexyl)phthalate (BEHP) in 1992 exceeded the USEPA chronic aquatic life standard. No surface water samples from 1 m or less were collected, so samples taken at depths of 6 and 12 m were used for BEHP comparisons. BEHP was not analyzed in 1999.

Stressors of Concern

Stressors in Onondaga Lake generally exceeded guidelines (when available) or background levels (see Section 8.1 and Appendix B). Chloride, ammonia, nitrite, phosphorus, and sulfide have consistently exceeded water quality criteria. Although lake salinity has dropped to 1.1 ppt, this value is still an order-of-magnitude greater than the average world river salinity (0.1 ppt). Phosphorus and sulfide concentrations have also consistently exceeded the NYSDEC standards from 1992 to 2001. Narrative water quality standards (6 NYCRR Part 703.2) have been exceeded in the lake, specifically those for settleable solids (e.g., calcite), which may physically impact zooplankton.

10.3.3 Do Measured Concentrations of Chemicals and Stressors in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Sediment Concentrations to Sediment Guidelines

Concentrations of COCs/SOCs in sediments were used as a measurement endpoint to evaluate whether certain zooplankton life stages (e.g., eggs) that spend extended periods in contact with Onondaga Lake sediments could be adversely affected by chemicals and stressors.

Concentrations of COCs in sediments exceeded guidelines for all sediment COCs (i.e., antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, selenium, silver, vanadium, zinc, benzene, chlorobenzene, dichlorobenzenes [total], trichlorobenzenes [total], ethylbenzene, toluene, xylenes [total], hexachlorobenzene, total polycyclic aromatic hydrocarbons [PAHs], phenol, dibenzofurans, chlordanes, heptachlor/heptachlor epoxide, dieldrin, DDT and metabolites, total PCBs, and dioxins/furans).

The maximum surface sediment arsenic concentration of 47 mg/kg was detected in 2000 along the southwestern shore at Station S333. This value exceeded the NYSDEC and Ontario Ministry of the Environment (OME) lowest effect level (LEL) of 6 mg/kg, the National Oceanic and Atmospheric Administration (NOAA) effects range-low (ER-L) of 8.2 mg/kg, the USEPA toxic equivalent concentration (TEC) of 12 mg/kg, and the NYSDEC and OME severe effect level (SEL) of 33 mg/kg. Ten out of 19 samples (53 percent) analyzed for arsenic in 1992 and 59 out of 85 samples (69 percent) analyzed for arsenic in 2000 exceeded the site-specific probable effect concentration (PEC) of 2.4 mg/kg calculated for Onondaga Lake (see Section 10.5.3 and Chapter 9, Tables 9-13, 10-2, and 10-3).

The maximum surface sediment cadmium concentration of 15 mg/kg was detected in 2000 at Station S322, near the mouth of Ley Creek. This value exceeded the NYSDEC and OME LEL of 0.6 mg/kg, the USEPA TEC of 0.6 mg/kg, the NOAA ER-L of 1.2 mg/kg, the NYSDEC and OME SEL of 10 mg/kg, and the USEPA PEC of 11.7 mg/kg. Forty-five out of 114 samples (39 percent) analyzed for cadmium in 1992 and 23 out of 85 samples (27 percent) analyzed for cadmium in 2000 exceeded the site-specific PEC of 2.4 mg/kg calculated for Onondaga Lake.

The maximum surface sediment chromium concentration of 4,180 mg/kg was detected in 2000 at Station S327, near the mouth of Tributary 5A. This value exceeded the NYSDEC LEL and OME LEL of 26 mg/kg, the USEPA TEC of 56 mg/kg, the NOAA ER-L of 81 mg/kg, and the NYSDEC and OME SEL of 110 mg/kg, the USEPA PEC of 159 mg/kg, and the USEPA high no-effect concentration (NEC) of 312 mg/kg. Fifty-four out of 114 samples (47 percent) analyzed for chromium in 1992 and 40 out of 85 samples (47 percent) analyzed for chromium exceeded the site-specific PEC of 50 mg/kg calculated for Onondaga Lake.

The maximum surface sediment lead concentration of 750 mg/kg was detected in 2000 near the mouth of Harbor Brook (Station S352). This value exceeded the NYSDEC and OME LEL of 31 mg/kg, the USEPA TEC of 34 mg/kg, the NOAA ER-L of 47 mg/kg, USEPA NEC of 69 mg/kg, the NYSDEC SEL of 110 mg/kg, the OME SEL of 250 mg/kg, and the USEPA PEC of 396 mg/kg. Seventy out of 114 samples (61 percent) analyzed for lead in 1992 and 46 out of 85 samples (54 percent) analyzed for lead in 2000 exceeded the site-specific PEC of 35 mg/kg calculated for Onondaga Lake.

The maximum surface sediment mercury concentration of 78 mg/kg was detected in 2000 offshore from the East Flume outlet (Station S344). This value exceeded the NYSDEC LEL and NOAA ER-L of 0.15 mg/kg, the OME LEL of 0.2 mg/kg, the NYSDEC SEL of 1.3 mg/kg and the OME SEL of 2 mg/kg. Sixty out of 114 surface sediment samples (53 percent) analyzed for mercury in 1992 and 86 out of 157 samples analyzed for mercury in 2000 (55 percent) exceeded the site-specific PEC of 2.2 mg/kg calculated for Onondaga Lake.

The maximum surface sediment nickel concentration of 1,670 mg/kg was detected near the mouth of Tributary 5A (Station S327). This value exceeded the NYSDEC and OME LEL of 16 mg/kg, the NOAA ER-L of 21 mg/kg, the USEPA NEC of 38 mg/kg, the USEPA PEC of 39 mg/kg, the USEPA TEC of 40 mg/kg, the NYSDEC SEL of 50 mg/kg and the OME SEL of 75 mg/kg. Seventy-two out of 114 samples (63 percent) analyzed for nickel in 1992 and 50 out of 85 samples (59 percent) analyzed for nickel in 2000 exceeded the site-specific PEC of 16 mg/kg calculated for Onondaga Lake.

The maximum surface sediment dichlorobenzene (sum) concentration of 1,270 μg/gOC was detected in Onondaga Lake in 2000 offshore from the East Flume outlet (Station S344). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 12 μg/gOC and the acute toxicity criterion of 120 μg/gOC. Seventeen out of 114 samples (15 percent) analyzed for dichlorobenzenes in 1992 and 34 out of 85 (40 percent) analyzed for dichlorobenzenes in 2000 exceeded the site-specific PEC of 239 μg/kg calculated for Onondaga Lake.

The maximum surface sediment trichlorobenzene (sum) concentration of 261 μg/gOC was detected in 2000 offshore from the East Flume outlet (Station S344). This value exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 91 μg/gOC and the acute toxicity criterion of 910 μg/gOC. Three out of 114 samples (3 percent) analyzed for trichlorobenzenes in 1992 and 5 out of 85 samples (6 percent) analyzed for trichlorobenzenes in 2000 exceeded the site-specific PEC of 347 μg/kg calculated for Onondaga Lake.

The maximum surface sediment ethylbenzene concentration detected in the lake of 100 μg/gOC near Tributary 5A (Station S435) exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 24 μg/gOC, the NYSDEC acute toxicity criterion of 212 μg/gOC, the USEPA sediment quality benchmark (SQB) of 360 μg/gOC, and the Oak Ridge National Laboratory (ORNL) secondary chronic criterion of 8.9 μg/gOC. One out of 114 samples (< 1 percent) analyzed for ethylbenzene in 1992 and 26 out of 61 samples (43 percent) analyzed for ethylbenzene in 2000 exceeded the site-specific PEC of 176 μg/kg calculated for Onondaga Lake.

The maximum surface sediment toluene concentration of 261 μg/gOC was detected near East Flume (Station S345). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 49 μg/gOC, the acute toxicity criterion of 235 μg/gOC, the USEPA SQB of 67 μg/gOC, and the ORNL secondary chronic criterion of 5 μg/gOC. Seventeen out of 114 samples (15 percent) analyzed for toluene in 1992 and 26 out of 62 samples (42 percent) analyzed for toluene in 2000 exceeded the site-specific PEC of 42 μg/kg calculated for Onondaga Lake.

The maximum surface sediment xylene (sum) concentration of 1,000 μg/gOC near Tributary 5A (Station S435). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 92 μg/gOC, the acute toxicity criterion of 833 μg/gOC, the USEPA SQB of 2.5 μg/gOC, and the ORNL secondary chronic criterion of 16 μg/gOC. Three out of 114 samples (3 percent) analyzed for xylenes in 1992 and 18 out of 37 (49 percent) analyzed for xylenes in 2000 exceeded the site-specific PEC of 561 μg/kg calculated for Onondaga Lake.

The maximum surface sediment hexachlorobenzene concentration of 105 μg/gOC was detected near Harbor Brook (Station S314). This concentration exceeded the NYSDEC wildlife bioaccumulation sediment criterion of 12 μg/gOC, the OME LEL of 2 μg/gOC, and the OME SEL of 24 μg/gOC. Twelve out of 89 samples (13 percent) analyzed for hexachlorobenzene in 1992 and 27 out of 85 samples (32 percent) analyzed for hexachlorobenzene in 2000 exceeded the site-specific PEC of 16 μg/kg calculated for Onondaga Lake.

The maximum surface sediment of total polycyclic aromatic hydrocarbons (PAHs) concentration of 29,430,000 μg/kg was detected near Tributary 5A (Station S435). This concentration exceeded the NOAA ER-L of 4,000 μg/kg and numerous criteria for individual PAH compounds. Site-specific PECs were calculated for individual PAH compounds and ranged between 146 μg/kg for benzo(a)pyrene and 1,436 μg/kg for fluoranthene.

The maximum surface sediment phenol concentration of 9.0 μg/gOC was detected between East Flume and Harbor Brook (Station S349). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.5 μg/gOC and the ORNL secondary chronic criterion of 3.1 μg/gOC. No samples analyzed for phenol in 1992 and 11 out of 85 samples (13 percent) analyzed from phenol in 2000 exceeded the site-specific PEC of 45 μg/kg calculated for Onondaga Lake.

The maximum surface sediment dibenzofuran concentration of 92 μg/gOC was detected near Harbor Brook (Station S313). This concentration exceeded the ORNL secondary chronic criterion of 42 μg/gOC. Two out of 19 samples (11 percent) analyzed for dibenzofuran in 1992 and 13 out of 85 samples (15 percent) analyzed for dibenzofuran in 2000 exceeded the site-specific PEC of 372 μg/kg calculated for Onondaga Lake.

The maximum surface sediment chlordanes (sum) concentration of 0.4 μg/gOC was detected near Harbor Brook (Station S314). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.03 μg/gOC and the NYSDEC wildlife bioaccumulation criterion of 0.006. No samples analyzed for chlordanes in 1992 and 8 out of 84 samples (10 percent) analyzed for chlordanes in 2000 exceeded the site-specific PEC of 5.1 μg/kg calculated for Onondaga Lake.

The maximum surface sediment heptachlor/heptachlor epoxide (sum) concentration of 1.7 μg/gOC was detected near Harbor Brook (Station S314). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.01 μg/gOC, the acute toxicity criterion of 13 μg/gOC, and the NYSDEC wildlife bioaccumulation criterion of 0.03 μg/gOC. There were not enough data to calculate a site-specific PEC for heptachlor/heptachlor epoxide.

The maximum surface sediment DDT and metabolites (sum) concentration of 3.6 μg/gOC was detected near Harbor Brook(Station S313). This concentration exceeded the NYSDEC 4-4'-DDT benthic aquatic life chronic toxicity sediment criterion of 1 μg/gOC and the OME LEL of 0.8 μg/gOC. One out of 19 samples (5 percent) analyzed for DDT and metabolites in 1992 and 5 out of 84 samples (6 percent) collected in 2000 exceeded the site-specific PEC of 30 μg/kg calculated for Onondaga Lake.

The maximum surface sediment total PCB concentration of 91 μg/gOC was detected between Tributary 5A and the East Flume (Station S344). This concentration exceeded the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 19 μg/gOC, the NYSDEC wildlife bioaccumulation criterion of 1.4 μg/gOC, and the OME LEL of 7 μg/gOC. Fourteen out of 114 samples (13 percent) analyzed for PCBs in 1992 and 42 out of 115 samples (37 percent) analyzed for PCBs in 2000 exceeded the site-specific PEC of 295 μg/kg calculated for Onondaga Lake.

The maximum surface sediment total dioxins/furan concentration of 129 μg/gOC was detected near Ley Creek (Station S322). This concentration exceeded the NYSDEC wildlife bioaccumulation criterion of 0.0002 μg/gOC. There were not enough data to calculate a site-specific PEC for dioxins/furans.

Although no guidelines address stressors in sediments, the large quantities of ionic waste stressors (e.g., calcium carbonate) deposited on Onondaga Lake sediments may also be detrimental to zooplankton eggs deposited in the sediment.

10.4 Assessment Endpoint: Sustainability of a Terrestrial Plant Community That Can Serve as a Shelter and Food Source for Local Invertebrates and Wildlife

10.4.1 Does the Terrestrial Plant Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern?

Measurement Endpoint: Field Observation of Onondaga Lake Plant Communities

The terrestrial plant communities found around Onondaga Lake reflect the development and disposal of contamination that has occurred near the lake over the last two centuries. As this BERA concentrates on the aquatic communities of Onondaga Lake, limited data were collected to evaluate the effects of COCs/SOCs on terrestrial communities. Only obvious effects, such as the sparse vegetation found on the wastebeds, can be directly attributed to activities at Honeywell facilities (i.e., disposal of Solvay and other industrial wastes).

10.4.2 Do Measured Concentrations of Chemicals and Stressors in Soil Exceed Toxicity Values for Terrestrial Plants?

Measurement Endpoint: Comparison of Measured Soil Concentrations to Plant Screening Values

Barium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, and zinc were the 12 COCs selected to evaluate plant exposure risks (Chapter 6, Table 6-1). Currently, there is no definitive guidance for setting terrestrial effect thresholds when conducting ecological risk assessments, and therefore the ORNL values used for screening (Efroymson et al., 1997a) were retained as plant toxicity reference values (TRVs). Soil concentrations were analyzed in the dredge spoils area and the wetland areas. Each of the four wetlands (SYW-6, SYW-10, SYW-12, and SYW-19) were evaluated individually for better characterization of the lake and because at least one of the wetlands, SYW-19 in the East Flume/Harbor Brook area, has been contaminated by Honeywell activities.

All COCs, except for copper, exceeded a hazard quotient (HQ) of 1.0 at one or more locations (Table 10-1). Mean and 95 percent upper confidence limit (UCL) concentrations of chromium, mercury, and vanadium exceeded an HQ of 1.0 at all locations (i.e., the dredge spoils and at each individual wetland). The highest HQs for mercury were seen at Wetland SYW-19 at the southwest corner of the lake near Harbor Brook, the highest chromium HQs were calculated for Wetland SYW-12 on the southeast corner of the lake near Ley Creek, and the highest vanadium HQs were calculated for Wetland SYW-10 on the western shore near Ninemile Creek. Zinc HQs exceeded 1.0 for all mean and 95 percent UCL concentrations at all wetland areas.

Selenium HQs exceeded 1.0 for mean and 95 percent UCL concentrations at the dredge spoils area and Wetland SYW-19. Selenium ratios of 1.0 were also exceeded for the 95 percent UCL for combined wetlands and Wetland SYW-10.

Lead HQs exceeded 1.0 for both mean and 95 percent UCL concentrations at all wetland locations, but were below 1.0 at the dredge spoils area. Nickel HQs exceeded 1.0 for the 95 percent UCL concentrations at all individual wetlands and for the mean concentration at Wetland SYW-19.

Cadmium HQs exceeded 1.0 for the 95 percent UCL concentrations for combined wetlands, Wetland SYW-6, and Wetland SYW-12. The mean exposure concentration also exceeded the soil benchmark at Wetland SYW-12. Thallium 95 percent UCL concentrations exceeded the soil benchmark at Wetland SYW-6 and SYW-10. The mean exposure concentration for thallium also exceeded the benchmark at Wetland SYW-10. Silver had a single exceedance of the benchmark at Wetland SYW-12 for the 95 percent UCL, and arsenic also had a single exceedance at Wetland SYW-10 for the 95 percent UCL.

The dredge spoils area had fewer exceedances than the four wetlands, which may be partially due to the absence of a hydrological connection to Onondaga Lake (i.e., the surface of the dredge spoils is approximately 10 ft above the elevation of the lake), and partially due to a soil cover that was placed over the contaminated spoils when they were constructed in 1966 to 1968 by filling in wetlands along the edge of the lake.

These results suggest the potential for adverse effects on plants via exposure to COCs in soils.

10.5 Assessment Endpoint: Sustainability of a Benthic Invertebrate Community That Can Serve as a Food Source for Local Fish and Wildlife

10.5.1 Does the Benthic Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern?

Measurement Endpoint: Analysis of Onondaga Lake Benthic Invertebrate Communities

The benthic macroinvertebrate community is closely associated with sediment and porewater, relying on these media for habitat, food, and exchange of gases. Therefore, the characteristics of the benthic invertebrate community are strongly affected by, and reflect, the quality of the sediment and water that the organisms inhabit. The overall health and structure of the benthic community can affect organisms such as fish and wildlife that depend upon it for food.

The Onondaga Lake benthic invertebrate community assessment investigated macroinvertebrate communities in areas of varying mixtures and concentrations of COCs/SOCs throughout the lake to create a general profile of community characteristics and determine whether ecologically based effects of COCs/SOCs could be inferred.

Of the 48 Onondaga Lake stations sampled in 1992, none were found to be non-impaired, 11 stations were found to be slightly impaired, 29 stations were found to be moderately impaired, and eight stations were found to be severely impaired (Figure 10-1). The severely impaired stations are primarily located at the southern end of the lake (i.e., between Onondaga Creek and Tributary 5A). One station (Station S68) considered to be severely impaired is located near Wastebeds 1 through 8. Moderately impaired stations are found throughout the lake.

NYSDEC's kick-sampling results from the mouths of the eight tributaries indicate that Harbor Brook, Ley Creek, Bloody Brook, Ninemile Creek, and Sawmill Creek are moderately impacted and that Onondaga Creek, the East Flume, and Tributary 5A are severely impacted. Based on sampling conducted by Honeywell in 1992, Tributary 5A, Harbor Brook, Onondaga Creek, and Ley Creek were classified as severely impaired, while the rest of the tributary mouth stations were moderately impaired (Figure 10-1).

All of the nine Onondaga Lake stations sampled in 2000 within the 5-m contour were found to be impaired to some extent (Figure 10-1). Two stations were found to be slightly impaired; six stations were found to be moderately impaired; and one station was found to be severely impaired. The severely impaired station (Station S317) is located in the southern end of the lake between the Metro outfall and the mouth of Onondaga Creek. Moderately impaired stations are found throughout the lake, clustered between Tributary 5A and Harbor Brook and near the mouths of Ninemile Creek and Ley Creek. Two of the three slightly impaired stations are located in Onondaga Lake: one (Station S365) is north of the mouth of Tributary 5A, and the other (Station S372) is in the northwestern portion of the lake.

The patterns described above and depicted in Figure 10-1 indicate that much of the littoral zone less than 5 m deep in Onondaga Lake and the mouths of the tributaries are impacted to some degree. The majority of moderately and severely impacted stations are found between Tributary 5A and Ley Creek. This coincides with the locations where most stations have three metrics that are significantly different than the Otisco Lake reference location. In addition, community-level measurements may be confounded by the influence of abiotic parameters (e.g., grain size and low DO levels) and the difficulty of distinguishing between directional (e.g., response to trend or gradient) and nondirectional (e.g., seasonal or annual) variability (Ingersoll et al., 1998).

10.5.2 Do Concentrations of Chemicals and Stressors in Sediment Influence Mortality, Growth, or Fecundity of Invertebrates Living In or On Lake Sediments?

Measurement Endpoint: Results of the 1992 and 2000 Sediment Toxicity Tests

Based on the 1992 toxicity tests, most amphipod toxicity was confined to an area in the southwestern corner of the lake, along Wastebeds 1 through 8 and along the Honeywell lakeshore area near Harbor Brook and the East Flume (Figure 10-2). Most chironomid toxicity was confined to the southern half of the lake, although toxicity was also found in two areas in the northern half of the lake (i.e., off Ninemile Creek and near Sawmill Creek). In the southern half of the lake, lethal chironomid toxicity was found in three general areas:

- Off Tributary 5A.

- Off Ley Creek.

- In the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume).

The results of the 42-day sediment toxicity tests from 2000 showed amphipod toxicity at six stations, including all of the shallow (i.e., < 5 m water depth) nearshore stations from Tributary 5A to the East Flume (Stations S332, S337, S342, S344, and S365) and near the Metro outfall (Station S317).

For the chironomid test conducted in 2000, lethal toxicity was found at nine stations, including all five of the shallow nearshore stations from Tributary 5A to the East Flume (i.e., Stations S332, S337, S342, S344, and S365), two stations off Ninemile Creek (Stations S302 and S303), and the stations off Ley Creek (Stations S320 and S323). In addition to the nine stations at which lethal toxicity was found for the chironomid test, sublethal toxicity was found at Station S317 off Onondaga Creek and at Station S372 along the northeastern shoreline of the lake. Chironomid emergence was affected at five locations in the southern portion of the lake at Stations S332, S337, S342, S344, and S354.

Overall, the results of the sediment toxicity tests confirmed that most sediment toxicity in Onondaga Lake is confined to the nearshore zone in the southern part of the lake between Tributary 5A and Ley Creek. By contrast, little toxicity is observed elsewhere in the lake, including the deeper parts of the entire lake and its eastern shore. The spatial patterns of amphipod and chironomid toxicity are presented in Figure 10-2.

10.5.3 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Values

Benthic macroinvertebrates are also exposed to COCs/SOCs in the water column. Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC and USEPA water quality values in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife value, but not the chronic water quality value for the protection of aquatic organisms.

Stressors in Onondaga Lake generally exceeded guidelines (when available) or background levels and are discussed in greater detail in Section 10.3.2. Chloride concentrations measured from April to November 1992 exceeded the USEPA chronic water quality criterion for chloride of 230 mg/L at all locations sampled (Chapter 8, Figure 8-8 and Appendix B, Table B-26). Although lake salinity has dropped from 3.6 ppt in 1981 to 1.1 ppt, it is still over an order-of-magnitude greater than the average world river salinity (0.1 ppt). From 1992 until 2001, phosphorus and sulfide concentrations also consistently exceeded the NYSDEC standards. Low levels of DO (Chapter 8, Figures 8-18 and 8-20) in the lake at depths greater than 3 m may also limit the benthic community.

The COC/SOC concentrations measured at both lake and tributary stations, as compared to water quality values, indicate that the benthic community may be adversely affected by the levels of COCs and stressors present in the water.

10.5.4 Do Measured Concentrations of Chemicals and Stressors in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Sediment Concentrations to Sediment Guidelines

Onondaga Lake sediment concentrations were compared to site-specific consensus PECs as another measurement endpoint in this strength-of-evidence approach. Consensus-based PECs for COCs in Onondaga Lake were developed to support an assessment to sediment-dwelling organisms and follow the methodology described in MacDonald et al. (2000) and Ingersoll et al. (2000). The PECs are the geometric mean of the apparent effects threshold (AET), probable effect level (PEL), threshold effects level (TEL), effects range-median (ER-M), and effects range-low (ER-L) sediment effect concentrations (SECs) presented in Chapter 9. In addition, the PECs:

- Provide a unifying synthesis of site-specific effects concentrations.
- Reflect causal rather than correlative effects.
- Account for the effects of sediment COCs.

The PECs do not consider the potential for:

- Bioaccumulation in aquatic species.
- Potential effects that could occur throughout the food web as a result of bioaccumulation.
- Synergistic or antagonistic effects of chemical mixes in the sediment.

Onondaga Lake PECs were developed for all compounds identified as COCs (see Chapter 6) based on the 1992 data. Measured surface sediment concentrations exceed the PEC consensus values at many locations throughout the lake. Table 10-2 presents exceedances at stations sampled in 1992 and Table 10-3 presents exceedances at stations sampled in 2000. Figure 10-3 shows the number of PEC exceedances for all of the identified COCs at each of the 1992 and 2000 stations. Only 14 sediment sampling locations in Onondaga Lake do not have at least one compound exceeding an HQ of 1.0 (i.e., sediment concentration less than the PEC). Many of the ratios of measured sediment concentrations to PECs exceed 10, or even 100, between Tributary 5A and Ley Creek. These total HQs are also presented in Tables 10-2 and 10-3. In addition, these sediment locations have the highest number of compounds – between 11 and over 30 compounds per sample – exceeding their PECs in a sample. Maps showing locations in the lake that exceed the SECs and PECs are included in Appendix F.

10.6 Assessment Endpoint: Sustainability of Local Fish Populations

10.6.1 What Does the Fish Community Structure Suggest about the Health of Local Fish Populations?

Measurement Endpoint: Comparison of Onondaga Lake Fish Communities to Reference Lakes

The current level of species diversity in Onondaga Lake is similar to values found in other New York State lakes, and growth rates, age distributions, and mortality rates of several species are similar to those observed in other northeastern US lakes (Auer et al., 1996a). However, sensitive species of fish, such as the Atlantic salmon (*Salmo salar*) and cisco (*Coregonus artedii*) that were historically present, are unable to survive in the lake. The dominant species in the lake are more pollution-tolerant and tolerant of warm-water conditions, unlike the historical cold-water fishery.

However, in contrast to comparison lakes, many of the species found in Onondaga Lake do not reproduce there and recruitment rates are unknown. Only 16 of 48 species captured in 1991 were found to reproduce in the lake, and reproduction within the lake varied by location. Many areas of Onondaga Lake are not suitable for fish reproduction due to industrial pollution and its effects on the lake ecosystem.

The composition of the fish community in the lake varies seasonally, with migration between the Seneca River and the lake being an important contributor to the variability. Several species of fish found in Onondaga Lake generally retreat to deeper cooler waters during hot weather. These are to a great extent the same fish species that migrate out of the lake. This suggests that after stratification the DO in the hypolimnion starts to decrease while the temperature of the epilimnion increases. In mid- to late-summer the water temperature of the lake reaches its highest level in the epilimnion and DO reaches its lowest level in the hypolimnion. Even before fall turnover (which lowers the overall lake DO), some species of fish can seek deeper, cooler waters. When they are unable to use the deeper part of the lake due to low DO, these species can move out of the lake to avoid the heat, particularly in late summer and early fall.

The limited fish reproduction in the lake and migration out of the lake during the fall indicate that Onondaga Lake alone cannot support the full diversity of the current fish community. Only with immigration into Onondaga Lake and refugia used during times of stress is the current diversity of the fish community sustainable.

10.6.2 Has the Presence of Chemicals and/or Stressors Influenced Fish Foraging or Nesting Activities?

Measurement Endpoint: Observations of Suitable Nesting Habitat and Populations of Juveniles

Fish reproduction within the lake varies by location. Based on the absence of juveniles in the catches of shoreline seine hauls, it is doubtful that species such as the walleye (*Stizostedion vitreum*) and northern pike (*Esox lucius*) reproduce in the lake. A lack of nursery area and adequate spawning sites has reduced successful reproduction of fish, resulting in poor year classes (Madsen et al., 1998). Spawning habitats constructed to improve fish habitat in Onondaga Lake had five to 20 times more fish nests than unmanipulated areas (Madsen et al., 1998). Lack of refugia in the deeper waters of the lake during portions of the year can also contribute to the low success or absence of reproduction of some species of the fish community in the lake.

Decreased water clarity, calcium carbonate precipitation, and increased salinity have reduced littoral zone vegetation (see Section 10.1), a critical area for young-of-year (YOY) fish. Areas characterized by the presence of aquatic macrophytes and submerged structures (e.g., near the lake outlet) supported the largest populations of juveniles. Areas with heavy silt loads and that are unprotected from wind are undesirable as spawning areas, as silt loads or wave action may cause eggs to be covered or removed from optimal areas.

Stressors, such as calcite and high salinity, have altered the phytoplankton and zooplankton communities in the lake, thereby affecting the food supply of many fish species. The low amount of littoral zone vegetation also results in lower biomass of macroinvertebrates and zooplankton, which serve as primary food for many YOY fish (Madsen et al., 1998).

The effects of industrial waste on Onondaga Lake have adversely affected fish reproduction and growth, as evidenced by low reproduction in the lake and fewer YOY fish than observed in similar lakes where the habitat has not been impacted by industrial contaminants, as is the case in Onondaga Lake.

10.6.3 Do Fish Found in Onondaga Lake Show Reduced Growth or Increased Incidence of Disease (e.g., Tumors, Lesions) as Compared to Fish from Other Lakes?

Measurement Endpoint: Observations of Incidence of Disease in Onondaga Lake Fish

Limited data are available regarding the incidence of disease in Onondaga Lake fish. During the 1992 nearshore fish study (PTI, 1993c), six fish (three banded killifish [*Fundulus diaphanus*], two pumpkinseed [*Lepomis gibbosus*], and one bluegill [*Lepomis macrochirus*]) were observed with abnormalities. Approximately 5,000 fish were collected during the study, but the number of fish examined was not specified in the report. In 1998, several kinds of grossly visible abnormalities were observed on three white suckers (*Catostomus commersoni*) during field sampling for the Geddes Brook/Ninemile Creek Remedial

Investigation (Exponent, 2001e). A total of 50 fish were collected in Geddes Brook/Ninemile Creek for analysis in 1998.

NYSDEC has not conducted any systematic observations for the fish from Onondaga Lake (Sloan, pers. comm., 2002). Seneca River fish were examined by Ringler et al. (Auer et al., 1996a) for external lesions and parasitic infestations that may be linked to industrial pollution. Rates of parasite occurrence and lesions in the Seneca River fish were determined to be at or below expected rates. Since levels of chemicals and stressors in the Seneca River are much lower than in Onondaga Lake, the relevance of this observation to the Onondaga Lake fish community is unknown.

10.6.4 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: **Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance and Qualitative Evaluation of Narrative Standards**

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC and USEPA water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife value, but not the chronic water quality values for the protection of aquatic organisms.

Stressors in Onondaga Lake generally exceed guidelines (when available) or background levels and are discussed in greater detail in Section 10.3.2. Chloride concentrations measured from April to November 1992 exceeded the USEPA chronic water-quality criterion for chloride of 230 mg/L at all locations sampled (Chapter 8, Figure 8-8 and Appendix B, Table B-26). Although lake salinity dropped to 1.1 ppt, this value is still an order-of-magnitude greater than the average world river salinity (0.1 ppt). From 1992 to 2001, phosphorus and sulfide concentrations have also consistently exceeded the NYSDEC standards.

The summed concentration of total ammonia and nitrate has continuously exceeded standards to protect non-salmonid (as well as salmonid) fish. Low levels of oxygen may also limit the fish community, particularly in fall. Fish generally move out of the lake during periods of low DO, as discussed previously (Auer et al., 1996a; Tango and Ringler, 1996).

Narrative water quality standards for turbidity and suspended, and settleable solids are also exceeded due to calcite resuspension, deposition, and formation of oncolites which result in impairment of the fish population in the lake (see Sections 10.1.3 and 10.6.2)

10.6.5 Do Measured Concentrations of Chemicals and Stressors in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Sediment Concentrations to Sediment Guidelines

The comparison of sediment concentrations to sediment criteria/guidance is only applicable to benthic-dwelling species of fish (e.g., catfish and carp) that are in close contact with sediments. Selected COCs detected in lake sediment in 1992 and 2000 were compared to NYSDEC sediment quality values in Appendix E (see Chapter 5, Table 5-5 for summary) and site-specific SECs and PECs in Appendix F.

Onondaga Lake PECs were developed for all compounds identified as COCs in Chapter 6 based on the 1992 data. Measured sediment concentrations exceed the PEC consensus values at many locations throughout the lake (Tables 10-2 and 10-3 and Figure 10-3). Only 15 sediment sampling locations in Onondaga Lake do not have at least one compound exceeding an HQ of 1.0. Many of the ratios of measured sediment concentrations to PECs exceed 10, or even 100, between Tributary 5A and Ley Creek. In addition, these sediment locations have the highest number of compounds – between 11 and over 30 compounds per sample – exceeding their PECs in a sample. Further discussion of COCs in sediments can be found in Section 10.3.3.

10.6.6 Do Measured Concentrations of Chemicals in Fish Exceed TRVs for Adverse Effects on Fish?

Measurement Endpoint: Comparison of Measured Fish Concentrations to Fish TRVs

Fish COCs were evaluated on a species-specific basis, as discussed in the following paragraphs. Some species (e.g., gizzard shad [*Dorosoma cepedianum*] and largemouth bass [*Micropterus salmoides*]) were only analyzed for a limited number of contaminants and, therefore, all risks may not be represented below.

The results presented below suggest the potential for adverse effects on most fish species via exposure to COCs in water, sediment, and prey.

10.6.6.1 Bluegill

Concentrations of chromium, vanadium, and zinc exceeded no observable adverse effect level (NOAEL) and lowest observable adverse effect level (LOAEL) TRVs at both the 95 percent UCL and mean concentrations in the bluegill (*Lepomis macrochirus*) (Table 10-4). Mercury and selenium concentrations in bluegill exceeded all TRVs except the LOAEL at the mean concentration. The arsenic 95 percent UCL exceeded the NOAEL.

10.6.6.2 Gizzard Shad

Only methylmercury was measured in gizzard shad. The 95 percent UCL and mean concentrations were above the NOAEL TRV (Table 10-4).

10.6.6.3 Carp

Concentrations of chromium, mercury, selenium, vanadium, and zinc exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations (Table 10-4). Dioxin/furan (TEQ) and arsenic concentrations in carp (*Cyprinus carpio*) exceeded all TRVs except the LOAEL at the mean concentration. PCB concentrations in carp exceeded the NOAEL at the 95 percent UCL mean concentrations and the endrin 95 percent UCL concentration exceeded the NOAEL.

10.6.6.4 Catfish

Concentrations of mercury, vanadium, and zinc exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean catfish concentrations (Table 10-4). Chromium and selenium exceeded all TRVs except the LOAEL at the mean concentration. Total PCBs in catfish exceeded the NOAEL at both the 95 percent UCL and mean concentrations.

10.6.6.5 White Perch

Concentrations of mercury exceeded the NOAEL and LOAEL TRVs for the white perch (*Morone americana*) at both the 95 percent UCL and mean concentrations (Table 10-4). Concentrations of chromium, selenium, and total PCBs exceeded NOAEL TRVs at both the 95 percent UCL and mean concentrations.

10.6.6.6 Smallmouth Bass

Concentrations of mercury and vanadium exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations for smallmouth bass (*Micropterus dolomieui*) (Table 10-4). Arsenic, selenium, and zinc exceeded all TRVs, except the LOAEL at the mean concentration. The 95 percent UCL concentration of PCBs exceeded the NOAEL TRV.

10.6.6.7 Largemouth Bass

Only mercury, DDT, PCBs, and dioxins/furans were analyzed in largemouth bass. Mercury exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations (Table 10-4). Dioxins/furans (TEQ) exceeded the NOAEL TRV at the 95 percent UCL.

10.6.6.8 Walleye

Concentrations of mercury exceeded the NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations for the walleye (*Stizostedion vitreum*) (Table 10-4). NOAELs were exceeded for chromium and total PCBs at both 95 percent UCL and mean concentrations.

10.7 Assessment Endpoint: Sustainability of Local Amphibian and Reptile Populations

10.7.1 What Do the Available Field-Based Observations Suggest about the Health of Local Amphibian and Reptile Populations?

Measurement Endpoint: Field Surveys of Local Amphibian and Reptile Populations

A field survey of Onondaga Lake found that habitats around the lake differed dramatically in the amphibian and reptile species found, with the lake itself and many other areas nearly devoid of herpetofauna (Ducey and Newman, 1995). The investigators concluded that the herpetofauna around the lake was generally depauperate, and were surprised by the absence of some common species. They found that the seven amphibian and six reptilian species found around the lake were considerably fewer than the 19 amphibian and 15 reptilian species recorded for Onondaga County as a whole during 1990 to 1996 by NYSDEC (1997b).

10.7.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC and USEPA water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife value, but not the chronic water quality value for the protection of aquatic organisms.

Stressors in Onondaga Lake generally exceeded guidelines (when available) or background levels and are discussed in greater detail in Section 10.3.2. Chloride concentrations measured from April to November 1992 exceeded USEPA chronic water-quality criterion for chloride of 230 mg/L at all locations sampled (Chapter 8, Figure 8-8 and Appendix B, Table B-26). Although lake salinity dropped to 1.1 ppt, this value is still an order-of-magnitude greater than the average world river salinity (0.1 ppt). From 1992 to 2001, phosphorus and sulfide concentrations have also consistently exceeded the NYSDEC standard.

10.7.3 Have Laboratory Studies Indicated the Potential for Adverse Effects to Amphibian Embryos from Exposure to Onondaga Lake Water?

Measurement Endpoint: Laboratory Toxicity Studies Using Onondaga Lake Surface Water

Ducey et al. (2000) directly assessed the toxicity of water from Onondaga Lake and associated wetlands on developing amphibian embryos. They found that water from connected wetlands and the lake has variable, but consistently negative, effects on amphibian development relative to controls. They hypothesized that there is a chemical interaction that affects amphibian embryos, because unfiltered Onondaga Lake water is highly toxic to embryos. Filtered water is also toxic, but to a lesser degree.

10.8 Assessment Endpoint: Sustainability of Local Insectivorous Bird Populations

10.8.1 Do Modeled Dietary Doses to Insectivorous Birds Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Insectivorous Bird Dietary Dose Concentrations to Toxicity Reference Values

Modeled dose concentrations of barium, chromium, methylmercury, mercury, selenium, and total PAHs for the tree swallow (*Tachycineta bicolor*) exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations (Table 10-5). Cadmium, lead, zinc, dichlorobenzenes, total PCBs, and dioxins/furans (TEQ) dose concentrations also exceeded the NOAEL at the 95 percent UCL and mean concentrations.

These results suggest the potential for adverse effects on insectivorous birds via exposure to COCs in water and prey.

10.8.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife protection value.

10.8.3 What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Bird Populations?

Measurement Endpoint: Field Observations of Insectivorous Birds

It is difficult to separate out the effects of chemical contamination on wildlife from those of development (i.e., habitat loss). Chapter 3, Table 3-11 lists bird species found in covertypes around Onondaga Lake. These covertypes support many insectivorous species, including swallows, mockingbirds, flycatchers, wrens, vireos, and warblers, among others. A number of species in these groups have been confirmed to breed around Onondaga Lake (NYSDEC, 2001a). However, field populations of insectivorous birds have not been studied. Without site-specific data on a representative insectivorous species, the significance of bird sightings is uncertain.

10.9 Assessment Endpoint: Sustainability of Local Benthivorous Waterfowl Populations

10.9.1 Do Modeled Dietary Doses to Benthivorous Waterfowl Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Benthivorous Bird Dietary Dose Concentrations to Toxicity Reference Values

Modeled dose concentrations of chromium and total PAHs for the mallard (*Anas platyrhynchos*) exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean dose concentrations (Table 10-6). Barium, methylmercury, and zinc also exceeded the NOAEL at both 95 percent UCL and mean dose concentrations and barium also exceeded the LOAEL at the 95 percent UCL concentration. Cadmium, dichlorobenzenes, and dioxins/furans (TEQ) dose concentrations exceeded the NOAEL at the 95 percent UCL concentration.

These results suggest the potential for adverse effects on waterfowl via exposure to COCs in water, sediment, and dietary sources.

10.9.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife protection value.

10.9.3 What Do the Available Field-Based Observations Suggest About the Health of Local Waterfowl Populations?

Measurement Endpoint: Field Observations of Local Waterfowl

Chapter 3, Table 3-11 lists bird species found in covertypes around Onondaga Lake, Table 3-12 lists additional species observed near the lake, and Table 3-13 lists waterfowl wintering near the lake. Onondaga Lake is home to many waterfowl, including ducks, geese, mergansers, scaups, loons, and grebes. However, few species of waterfowl are listed as confirmed or probable breeders in the New York State Breeding Bird Atlas (NYSDEC, 2001a). Although it is clear that Onondaga Lake is an important resource for many resident and migratory species of waterfowl, its significance as a breeding area is unknown and little can be inferred about the health of local waterfowl populations.

10.10 Assessment Endpoint: Sustainability of Local Piscivorous Bird Populations

10.10.1 Do Modeled Dietary Doses to Piscivorous Birds Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Piscivorous Bird Dietary Dose Concentrations to Toxicity Reference Values

Modeled methylmercury dose exposure concentrations exceeded NOAEL and LOAEL TRVs at both 95 percent UCL and mean concentrations for all three piscivorous birds modeled (belted kingfisher [*Ceryle alcyon*], great blue heron [*Ardea herodias*], and osprey [*Pandion haliaetus*]) (Tables 10-7 to 10-9). Methylmercury exceeded NOAELs for all piscivorous receptors by an order-of-magnitude. All modeled dose concentrations of DDT exceeded all TRVs for the belted kingfisher, while DDT NOAELs were exceeded for 95 percent UCL and mean dose concentrations for the great blue heron and osprey.

The total PAH and total PCB exposure dose concentrations were greater than the NOAELs for the belted kingfisher and great blue heron at both 95 percent UCL and mean concentrations. The 95 percent UCL total PAH concentration also exceeded the LOAEL for the belted kingfisher. Total PCBs exceeded the NOAEL at the 95 percent UCL concentration for the osprey.

Dioxins/furans (TEQ) exposure dose concentrations were greater than the NOAELs for the belted kingfisher at both 95 percent UCL and mean concentrations. Zinc exposure dose concentrations were greater than the NOAELs for the osprey and great blue heron at the 95 percent UCL concentration, and the mean zinc concentration for the osprey also exceeded the NOAEL.

These results suggest the potential for adverse effects on piscivorous birds via exposure to COCs in water, sediment, and dietary sources.

10.10.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Measurement Endpoint: Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife protection value.

10.10.3 What Do the Available Field-Based Observations Suggest About the Health of Local Piscivorous Bird Populations?

Measurement Endpoint: Field Observations of Local Piscivorous Birds

Chapter 3, Table 3-11 lists bird species found in covertypes around Onondaga Lake and Table 3-12 lists additional species observed near the lake. Onondaga Lake is home to a number of piscivorous bird species including kingfishers, herons, bald eagles, osprey, cormorants, gulls, and terns, some of which have been observed year-round near the lake (Kirkland Bird Club, 2002). The presence of these species indicates that suitable habitat is available in Onondaga Lake. However, very few piscivorous species are listed as confirmed or probable breeders in the New York State Breeding Bird Atlas (NYSDEC, 2001a). The mud flats area around the mouth of Ninemile Creek is considered by local birders to be a sensitive migratory area that provides good habitat for birds (Kirkland Bird Club, 2002).

10.11 Assessment Endpoint: Sustainability of Local Carnivorous Bird Populations

10.11.1 Do Modeled Dietary Doses to Carnivorous Birds Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Carnivorous Bird Dietary Dose Concentrations to Toxicity Reference Values

Modeled total PAH exposure dose concentrations exceeded NOAEL and LOAEL TRVs at both 95 percent UCL and mean concentrations for the red-tailed hawk (*Buteo jamaicensis*) (Table 10-10).

Modeled doses of dioxins/furans (TEQ) exceeded the NOAEL at both the 95 percent UCL and mean concentrations. The DDT NOAEL was exceeded for the 95 percent UCL concentration.

These results suggest the potential for adverse effects on carnivorous birds via exposure to COCs in water, sediment, and dietary sources.

10.11.2 What Do the Available Field-Based Observations Suggest About the Health of Local Carnivorous Bird Populations?

Measurement Endpoint: Field Observations of Local Carnivorous Birds

It is difficult to separate out the effects of chemical contamination on wildlife from those of habitat loss and development. Table 3-11 lists bird species found in covertypes around Onondaga Lake. The covertypes may support carnivorous species such as the turkey vulture (*Cathartes atratus),* red-tailed hawk, sharp-shinned hawk (*Accipiter striatus*), Cooper's hawk (*Accipter cooperii*), and American kestrel (*Falco sparverius*). A number of species in these groups have been confirmed to breed around Onondaga Lake (NYSDEC, 2001a). However, populations of carnivorous birds have not been studied at Onondaga Lake to place these observations into the proper perspective.

10.12 Assessment Endpoint: Sustainability of Local Insectivorous Mammal Populations

Insectivorous receptors around Onondaga Lake were divided into insectivores feeding on aquatic invertebrates and insectivores feeding on terrestrial invertebrates.

10.12.1 Do Modeled Dietary Doses to Insectivorous Mammals Feeding on Aquatic Invertebrates Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Mammal Insectivorous Dietary Dose Concentrations to Toxicity Reference Values

The little brown bat (*Myotis lucifugus*) was used as a representative receptor for mammals feeding on insects with an aquatic life phase. Modeled dose concentrations of barium, chromium, methylmercury, and total PAHs for the little brown bat exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean exposure concentrations (Table 10-11). Copper and dioxins/furans (TEQ) dose concentrations exceeded the NOAEL at the 95 percent UCL and the mean dose concentrations and the LOAEL at the 95 percent UCL concentration. Cadmium, vanadium, and hexachlorobenzene exceeded NOAELs at both 95 percent UCL and mean dose concentrations. Total xylenes exceeded the NOAEL and LOAEL at the 95 percent UCL concentration. Mercury and arsenic exceeded an HQ of 1.0 at the 95 percent UCL concentration based on the NOAEL TRV.

These results suggest the potential for adverse effects on insectivorous mammals via exposure to COCs in water and prey with an aquatic life-phase.

10.12.2 Do Modeled Dietary Doses to Insectivorous Mammals Feeding on Terrestrial Invertebrates Exceed Toxicity Reference Values for Adverse Reproductive Effects?

Measurement Endpoint: Comparison of Modeled Insectivorous Mammal Dietary Dose Concentrations to Toxicity Reference Values

Due to the small home range of the short-tailed shrew (*Blarina brevicauda*) (used as the representative receptor for insectivorous mammals feeding on terrestrial prey), each discretely sampled area was modeled individually for the four wetland areas (SYW-6, SYW-10, SYW-12, and SYW-19) and the dredge spoils area (Table 10-12).

Wetland SYW-19, along the southwest corner of the lake near the mouth of Harbor Brook, had the greatest number of exceedances (12), with modeled doses of methylmercury, total PAHs, hexachlorobenzene, and dioxins/furans (TEQ) exceeding LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Hazard quotients of total PAHs and dioxins/furans were up to three orders-of-magnitude above 1.0. NOAELs for arsenic, cadmium, lead, selenium, vanadium, trichlorobenzenes, and total PCBs were exceeded at both upper and mean dose exposures.

Wetland SYW-10 on the west side of the lake near the mouth of Ninemile Creek had 10 HQ exceedances. Modeled doses of methylmercury and total PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Arsenic, thallium, vanadium, hexachlorobenzene, and dioxins/furans (TEQ) NOAELs were exceeded at both upper and mean dose exposures, while the cadmium and lead NOAELs were exceeded at the 95 percent UCL dose.

Wetland SYW-6 at the northwest end of the lake also had ten HQ exceedances. Modeled doses of methylmercury and total PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Additional exceedances at SYW-6, were the NOAELs for arsenic, thallium, vanadium, and dioxins/furans at both upper and mean dose exposures, and the cadmium and dioxins/furans LOAEL was also exceeded at the 95 percent UCL dose. The NOAEL and LOAEL for selenium were exceeded at the 95 percent UCL and the chromium and lead NOAELs were exceeded at the 95 percent UCL dose.

Wetland SYW-12 at the southeast end of the lake had eight HQ exceedances. Modeled doses of methylmercury and total PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Cadmium and vanadium NOAELs were exceeded at both upper and mean dose exposures. Arsenic, lead, hexachlorobenzene, and dieldrin NOAELs were exceeded at the 95 percent UCL dose. Data for dioxins/furans were not available at SYW-12.

At the dredge spoils area, arsenic, vanadium, hexachlorobenzene, and total PAH NOAELs were exceeded at both upper and mean dose exposures for surface soils. The hexachlorobenzene 95 percent UCL dose also exceeded the LOAEL and the selenium NOAEL was exceeded at the 95 percent UCL dose.

These results suggest the potential for adverse effects on insectivorous mammals via exposure to COCs in water, sediment, and terrestrial prey. The potential for adverse effects was calculated to be greater in wetlands areas than the dredge spoils area (surface soils). The covertypes of the wetland areas provide more suitable habitat for wildlife than the dredge spoils area (Chapter 3).

10.12.3 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Measurement Endpoint: **Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance**

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife protection value.

10.12.4 What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Mammal Populations?

Measurement Endpoint: **Field Observations of Local Insectivorous Mammals**

It is difficult to separate out the effects of chemical contamination on wildlife from those of habitat loss and development. Chapter 3, Table 3-14 lists mammalian species found in covertypes around Onondaga Lake. Several insectivorous species, such as shrews and bats, are found in these covertypes. However, local populations of insectivorous mammals have not been studied to determine whether they have been impacted.

10.13 Assessment Endpoint: Sustainability of Local Piscivorous Mammal Populations

10.13.1 Do Modeled Dietary Doses to Piscivorous Mammals Exceed Toxicity Reference Values for Adverse Effects on Reproduction?

Measurement Endpoint: **Comparison of Modeled Piscivorous Mammal Dietary Dose Concentrations to Toxicity Reference Values**

Modeled dose concentrations of total PCBs in the mink (*Mustela vison*) exceeded the NOAEL and LOAEL TRVs at both the 95 percent UCL and mean exposure doses (Table 10-13). Modeled mink dietary doses of methylmercury, total PAHs and dioxins/furans exceeded NOAELs at the 95 percent UCL and mean exposure dose levels and the LOAEL at the 95 percent UCL (Table 10-13). Hexachlorobenzene exceeded a HQ of 1.0 when compared to the NOAEL at both 95 percent UCL and mean concentrations for the mink.

Modeled dose concentrations of methylmercury and total PCBs in the river otter (*Lutra canadensis*) exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean exposure doses (Table 10-14). Modeled river otter dietary doses of total PAHs, DDT and metabolites, and dioxins/furans exceeded NOAELs at the 95 percent UCL and mean exposure dose levels (Table 10-14). DDT and metabolites also exceeded the LOAEL at the 95 percent UCL concentration for the river otter.

These results suggest the potential for adverse effects on piscivorous mammals via exposure to COCs in water, sediment, and prey.

10.13.2 Do Measured Concentrations of Chemicals and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Measurement Endpoint: **Comparison of Measured Surface Water Concentrations to Water Quality Standards, Criteria, and Guidance**

Selected COCs detected in lake surface water in 1992 and 1999 were compared to NYSDEC water quality standards, criteria, and guidance in Appendix B (see Chapter 5, Table 5-3 for summary). With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate), exceeded USEPA chronic aquatic or Tier II water quality values (Chapter 4, Table 4-4). Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the wildlife protection value.

10.13.3 What Do the Available Field-Based Observations Suggest About the Health of Local Piscivorous Mammal Populations?

Measurement Endpoint: Field Observations of Local Piscivorous Mammals

It is difficult to separate out the effects of chemical contamination on wildlife from those of habitat loss and development. Chapter 3, Table 3-14 lists mammalian species found in covertypes around Onondaga Lake. The mink and river otter are the only primarily piscivorous mammals found in these covertypes. The New York River Otter Project has records of river otters from the Onondaga Lake area from 1994 to 2002 (NYSDEC, 2002a). Otters have been observed at several locations along Ninemile Creek (e.g., Erie Canal, NY State Fairgrounds) and Onondaga Creek. The low numbers of mink and river otter sighted around Onondaga Lake may be due to a number of factors, such as inadequate habitat, disturbance by humans, and chemical contamination, among others. There has been no standardized effort to document mink and otter populations.

10.14 Summary

Multiple lines of evidence were used to evaluate major components of the Onondaga Lake ecosystem to determine if lake contamination has adversely affected plants and animals around Onondaga Lake. Almost all lines of evidence indicate that the Honeywell-related contaminants, including ionic waste in Onondaga Lake, have produced adverse ecological effects at all trophic levels examined.

The aquatic macrophytes in the lake have been adversely affected by lake conditions, and the resulting loss of macrophyte habitat that formerly provided valuable feeding and nursery areas has undoubtably affected the aquatic invertebrates and vertebrates living in Onondaga Lake. In addition to general habitat loss, there has been bioaccumulation of mercury and possibly other contaminants in most organisms serving as a food source in the lake, including phytoplankton, zooplankton, benthic invertebrates, and fish.

Site-specific sediment PECs indicate adverse effects in the southern areas of the lake and near Ninemile Creek. The area near the Honeywell sites between Tributary 5A and Harbor Brook exhibited the greatest number of exceedances of the PECs (Figure 10-3).

Comparisons of measured tissue concentrations and modeled doses of contaminants to TRVs show exceedances of HQs for site-related chemicals throughout the range of the point estimates of risk. Many of the contaminants in the lake are persistent and, therefore, the risks associated with these contaminants are unlikely to decrease significantly in the short term in the absence of remediation.

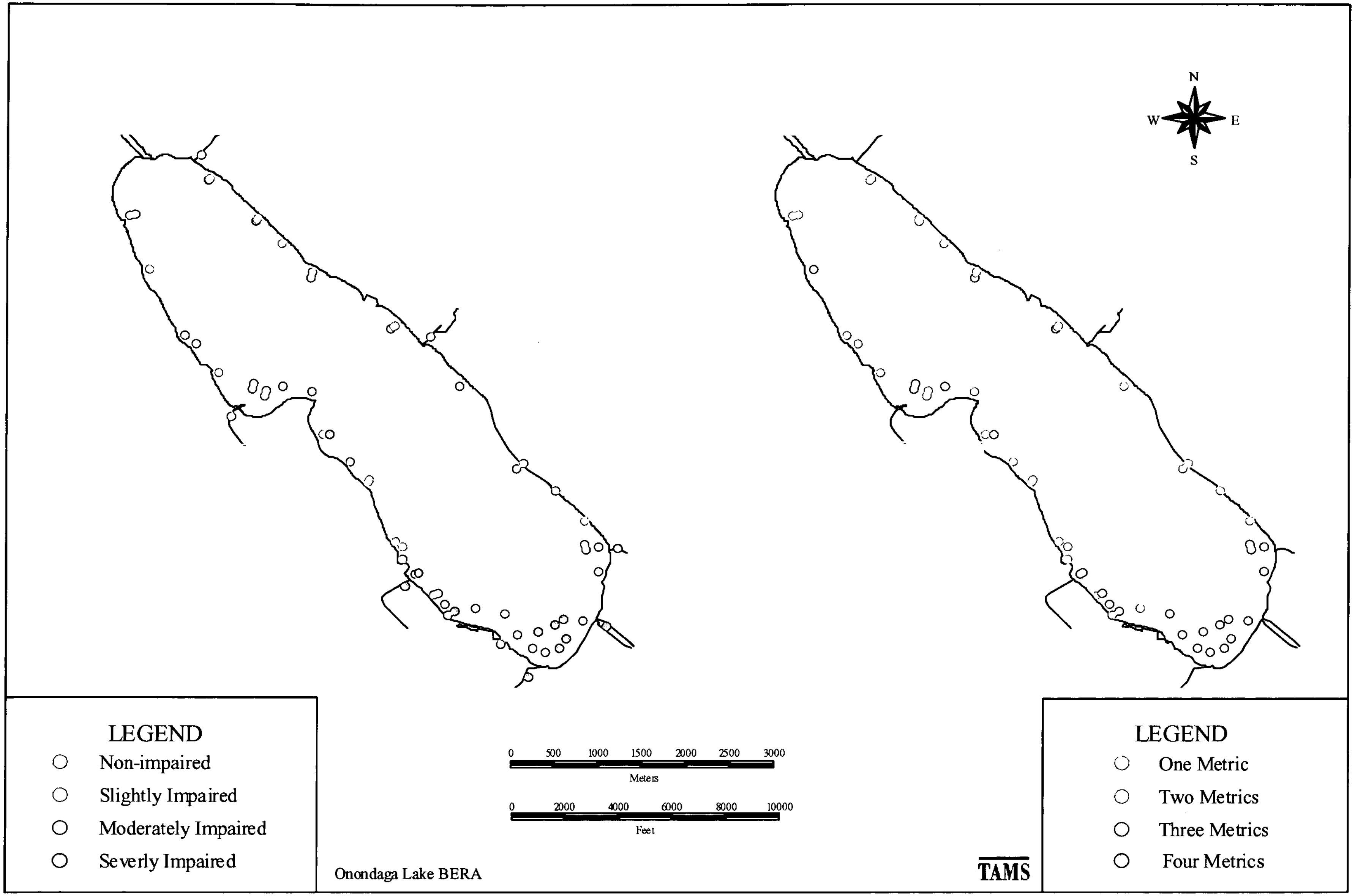


Figure 10-1
Comparison of Benthic Community Metrics and Statistical Analyses in 1992 and 2000

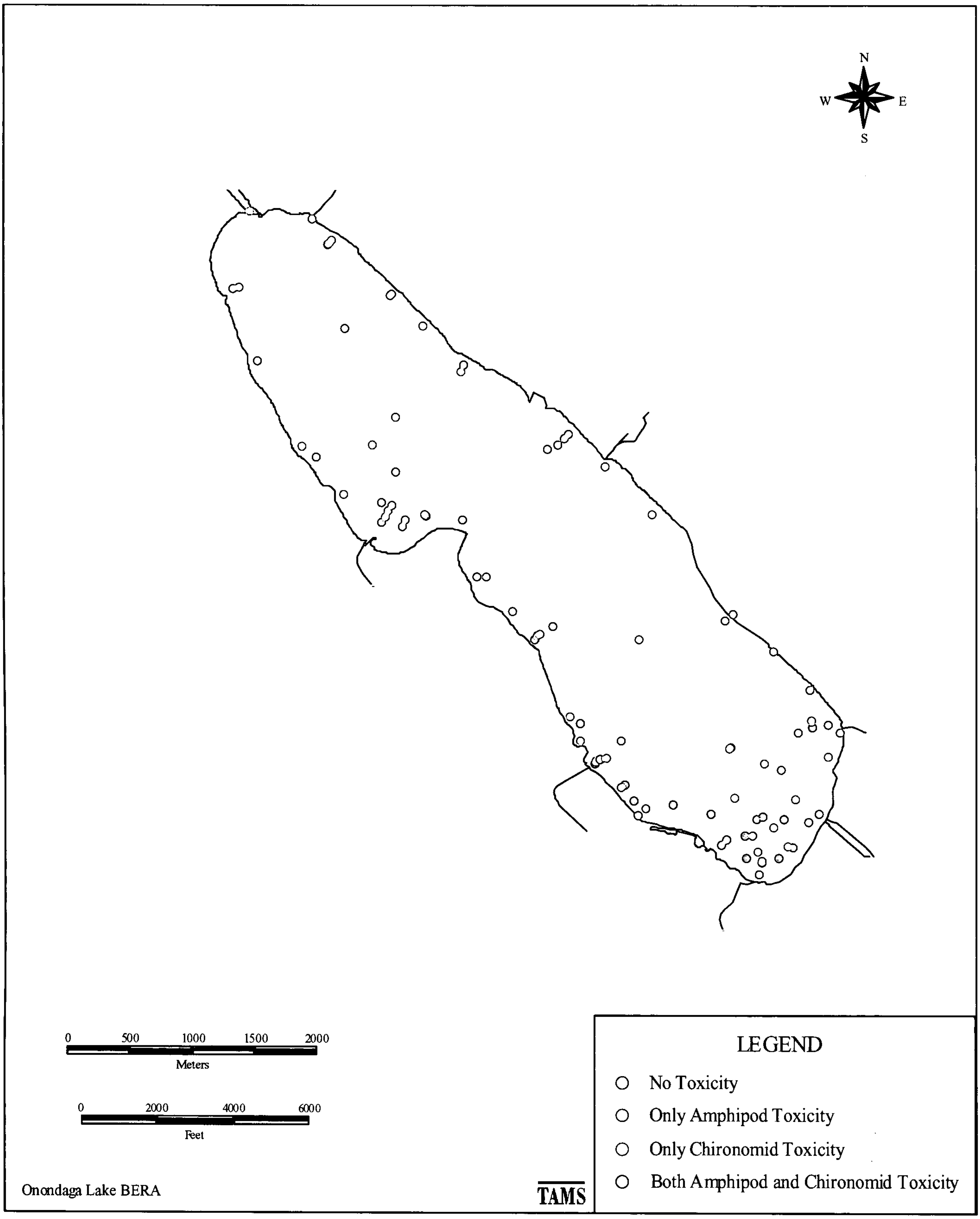


Figure 10-2
Spatial Patterns of Amphipod and Chironomid Toxicity in 1992 and 2000

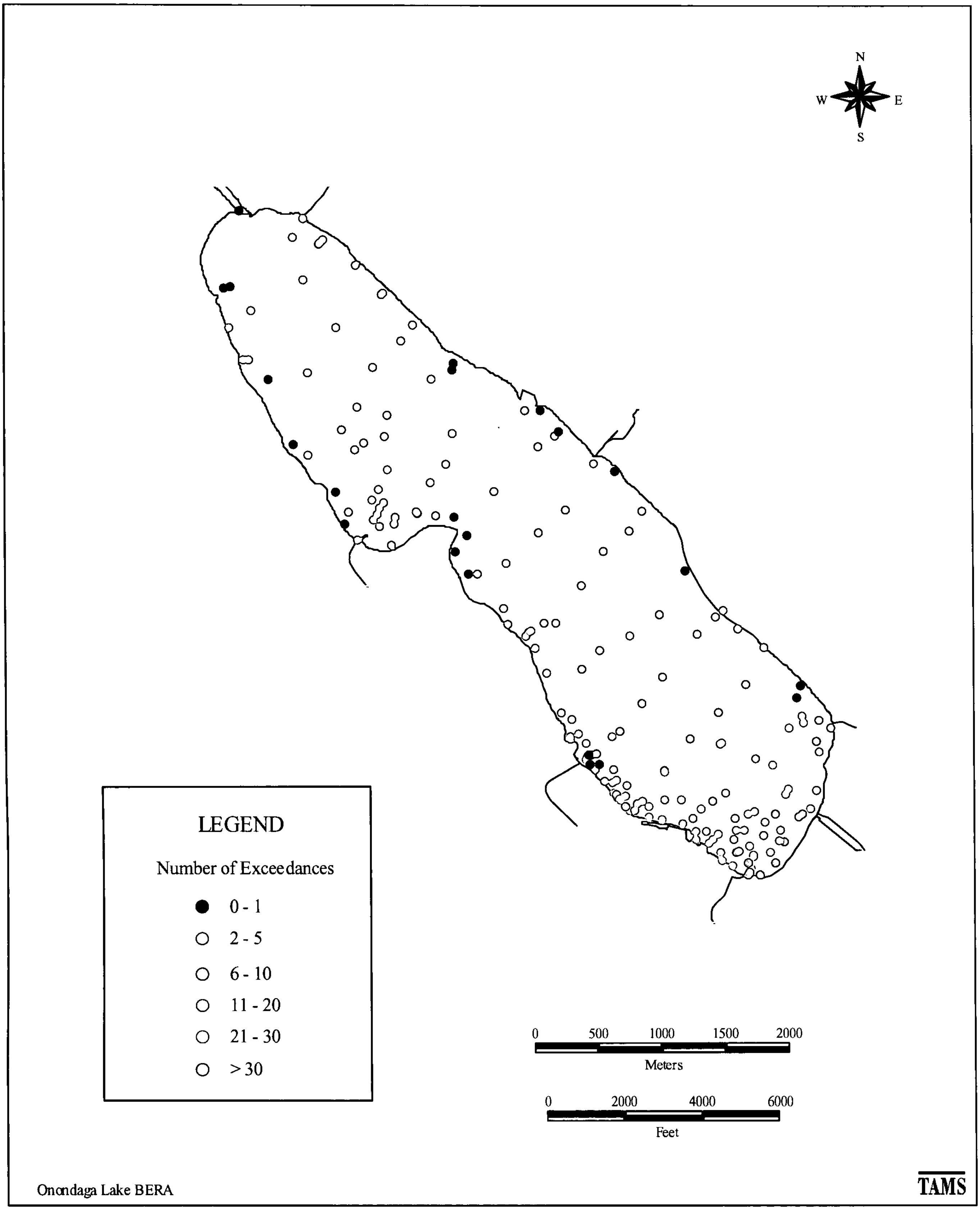


Figure 10-3
Sediment Locations Where Consensus Based Values are Exceeded

Table 10-1. Ratios of COCs in Soils Near Onondaga Lake in 2000 to ORNL Soil Benchmarks for Plants

COC	Combined Wetlands 95%UCL HQ	Combined Wetlands Mean HQ	SYW-6 95%UCL HQ	SYW-6 Mean HQ	SYW-10 95%UCL HQ	SYW-10 Mean HQ	SYW-12 95%UCL HQ	SYW-12 Mean HQ	SYW-19 95%UCL HQ	SYW-19 Mean HQ	Dredge spoils 95%UCL HQ	Dredge spoils Mean HQ
Arsenic	0.6	0.5	0.6	0.3	1.8	0.7	0.4	0.3	0.9	0.7	0.8	0.5
Cadmium	3.6	0.5	3.6	0.9	0.2	0.1	2.2	1.3	0.6	0.3	0.0 *	0.0 *
Chromium	51	39	154	49	47	27	115	66	54.6	42.8	29	17
Copper	4.4E-02	4.0E-02	0.1	3.6E-02	0.1	0.0	4.7E-02	3.0E-02	0.1	0.0	4.8E-02	4.2E-02
Lead	2.1	1.2	3.5	1.4	2.3	1.2	2.3	1.5	5.2	2.4	0.3	0.2
Mercury	62	9.9	15	4.3	11	7.0	5.0	2.2	82.7	48.8	13	2.2
Nickel	0.9	0.8	2.1	1.0	1.1	0.7	1.1	0.6	1.5	1.1	0.6	0.5
Silver	0.4	0.2	0.7	0.3	0.0 *	0.0 *	1.4	0.6	0.6	0.2	0.0 *	0.0 *
Selenium	1.2	0.9	2.5	0.8	1.8	0.7	0.9	0.4	1.7	1.4	1.4	1.0
Thallium	0.8	0.6	1.4	0.6	2.5	1.5	0.0 *	0.0 *	0.0 *	0.0 *	0.0 *	0.0 *
Vanadium	8.2	7.2	11	6.5	15	7.8	7.8	4.3	6.5	6.2	14	9.5
Zinc	3.2	2.4	10	3.6	2.4	1.9	4.8	3.2	2.8	2.3	1.0	0.8

Notes:

Hazard quotients equal to or greater than one are outlined.

*- Denotes all ND samples

Table 10-2. Summary of Ratios to Consensus PECs for COC Concentrations in Surface Sediments in Onondaga Lake in 1992[a,b]

Chemical	Consensus PECs	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16
Metals	**mg/kg**																
Antimony	3.6										1.77					0.86	1.10
Arsenic	2.4	1.19					0.26				0.67					1.55	0.59
Cadmium	2.4	0.36	2.95	1.69	1.01	2.4	0.46	0.46	1.18	1.18	1.43	0.20	1.81	1.26	0.59	1.05	0.51
Chromium	50.3	0.59	2.76	1.67	1.31	1.69	0.39	0.37	1.65	1.41	0.95	0.12	1.07	0.66	0.59	0.95	0.66
Copper	32.9	1.54	4.34	1.91	1.83	2.55	1.74	1.53	2.54	2.54	1.56	0.26	1.92	2.51	0.89	1.49	1.07
Lead	34.5	2.27	4.61	3.51	2.58	4.8	1.62	2.0	2.9	2.1	1.72	0.45	2.42	3.10	2.89	4.20	0.78
Manganese	278.3	0.64					0.34				0.78					0.73	0.86
Mercury	2.2	2.25	8.21	8.35	9.25	5.06	0.50	1.45	3.58	2.95	0.77	0.24	1.27	0.50	13.58	31.25	0.11
Nickel	16.4	0.94	4.67	2.93	1.36	1.55	0.45	0.45	1.37	0.96	0.71	0.26	1.18	1.28	1.83	1.78	0.27
Selenium	0.6	0.63					0.43									0.74	0.45
Silver	1.3	0.49									0.74					1.79	0.66
Vanadium	5.6	1.00					0.21				0.46					0.45	0.09
Zinc	88.2	1.07	3.13	2.11	1.56	2.20	0.49	0.88	2.06	2.01	1.36	0.30	1.79	2.47	0.71	1.07	0.74
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4	4.25	2.19	0.34	0.20					0.18							
Chlorobenzene	428.4	23.36	7.70	2.57	1.35	0.49			0.21	0.51	0.02		0.04		9.10	100	
Dichlorobenzenes (Sum)	238.6	5.76	17.48	3.81	2.67	0.73	0.57	0.02	0.44	0.70					77.1	95.6	
Ethylbenzene	175.7																
Toluene	41.8	10.60	18.90	0.41	0.69	0.53		0.65	0.33	0.48		5.50		3.11	100		
Trichlorobenzenes (Sum)	346.5	0.83	1.62	0.07			0.12								12.12		
Xylene (Total)	560.8	1.08												0.86	4.99		
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0	3.49					0.12				1.26						
Hexachlorobenzene	16.4		7.94	17.70	5.07	7.33		0.67	3.97	1.28			1.71	0.28	10.38		
Phenol	45.0																
Naphthalene	917.4	2.29	3.16	1.20	1.96	4.58	0.09		0.93	0.58	0.75		6.54	12.0	32.7	25.1	
Acenaphthene	860.7	1.18	1.28			3.37	0.22	17.4	0.91		1.03		8.60				0.05
Fluorene	264.3	4.67	2.16	2.27	1.44	5.30	0.61	9.8	0.36	0.79	13.24		3.78	27.24			0.27
Phenanthrene	542.7	10.38	0.88	0.88	0.55	3.87	1.36	11.6	1.00	0.39	29.5		6.08	40.54	12.72	4.42	1.00
Anthracene	206.7	15.97	1.89	3.29	1.02	6.3	2.37	19.4	0.82	0.28	21.3	2.76	5.81	43.06	23.23		1.16
Pyrene	343.8	21.72	5.24	7.85	2.15	11.6	3.49	27.6	2.76	1.51	49.5	1.11	19.5	9.31			3.20
Benz(a)anthracene	191.5	21.59	4.02	4.81	1.78	10.4	4.07	22.5	2.56	1.25	40.2	1.78	19.3	13.58	5.22		3.29
Chrysene	253.2	18.17	3.36	3.44	1.66	11.8	3.12	18.2	3.79	1.07	29.6		28.4				2.65
Benzo(b)fluoranthene	908.4	6.90	2.20	2.97	2.42	4.6	1.43	4.6	0.83	0.64	10.7	1.43	5.83	31.93	1.54		1.21
Benzo(a)pyrene	146.4	25	3.2	3.3	2.3	10.9	5.9	9.6	1.4	0.9	53.3	4.8	17.1	18.4			0.51
Indeno(1,2,3-cd)pyrene	182.9	10.57	6.01	2.62	2.24	4.26	1.75		1.97	0.71	10.4						1.53
Dibenz(a,h)anthracene	157.2	4.24	2.61			4.07	0.76		1.27	0.37	4.64	2.35	4.52	5.66			0.64

Table 10-2. (cont.)

Chemical	Consensus PECs	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16
Benzo(g,h,i)perylene	779.7	**2.18**	**2.69**	**3.98**	**2.05**	**5.64**	0.37	**4.62**	**1.67**	0.71	**2.44**	**2.82**	**6.28**	**11.9**	**3.85**		0.33
Acenaphthylene	1,300.7	0.49				**2.00**	0.09			0.92	**1.54**		**4.84**				0.06
Benzo(k)fluoranthene	202.5			**2.62**		**4.89**		**12.35**	0.94	0.94		**4.00**	**10.4**	**18.8**			
Fluoranthene	1,436.3	**4.62**	**6.68**		**1.53**	**3.41**	0.63	**59.88**	0.91	**1.25**	**6.96**	**18.1**	**10.4**	**160**	**27.85**	**1.60**	0.65
Pesticides/Polychlorinated Biphenyls	**µg/kg**																
Chlordane (Sum)	5.1	1.0															
DDT and metabolites	29.6	0.55									0.25						
Aroclor-1016	110.9															**1.62**	
Aroclor-1248	203.7	0.82	**3.29**	**3.68**	**2.01**	**2.16**		0.29	**1.47**	**1.03**			**1.42**	**1.42**	**1.18**		0.83
Aroclor-1254	76.1																
Aroclor-1260	163.6		**2.32**	**5.50**	**4.03**	**1.47**		0.23	**1.10**	0.73			**3.42**	0.43		**1.34**	
PCBs (Sum)	294.8	0.84	**3.56**	**5.60**	**3.63**	**2.31**		0.33	**1.63**	**1.12**			**2.88**	**1.22**	0.95	**1.36**	0.67
Exceedances		25	29	24	24	27	10	15	18	13	20	10	26	20	17	16	8
Total Hazard Index		**207**	**140**	**99**	**58**	**131**	**27**	**223**	**39**	**21**	**283**	**45**	**178**	**409**	**341**	**276**	**15**

Table 10-2. (cont.)

Chemical	Consensus PECs	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32
Metals	**mg/kg**																
Antimony	3.6																
Arsenic	2.4				2.26		1.21					2.26					
Cadmium	2.4	1.60	3.62	1.98		0.6	1.47	1.26	1.64	1.43		1.26	1.26	1.73	1.01	1.26	1.18
Chromium	50.3	0.98	2.11	1.37	0.32	0.50	2.05	1.81	1.69	1.32	0.47	1.44	39.5	7.73	1.49	1.31	1.25
Copper	32.9	0.72	1.69	2.15	0.93	1.12	2.87	2.09	2.40	2.47	0.21	2.41	5.26	2.73	1.76	1.96	1.95
Lead	34.5	0.45	1.63	2.39	1.48	3.4	2.98	2.4	2.5	2.3	0.14	2.29	6.90	2.69	2.03	2.50	2.27
Manganese	278.3				0.78		0.88					1.56					
Mercury	2.2	0.08	0.82	0.91	5.17	12.63	3.17	2.77	2.27	1.41	0.10	1.41	2.36	1.22	1.45	1.32	1.18
Nickel	16.4	0.61	1.55	1.44	1.22	1.76	1.72	1.31	1.39	1.26	0.38	1.88	39.70	4.39	1.80	1.50	1.81
Selenium	0.6						0.90					1.33					
Silver	1.3				0.69		3.98					1.64					
Vanadium	5.6				0.52		2.18					2.91					
Zinc	88.2	1.00	2.23	2.19	1.53	1.06	2.35	1.90	2.21	2.29	0.32	2.24	2.10	2.22	1.73	2.05	2.03
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4				4.99		0.09	0.37	0.20	0.11		0.20	37.91		0.36	0.15	0.13
Chlorobenzene	428.4				2.57	7.70	0.11	0.35	0.23	0.12		0.20	70.03	0.26	0.35	0.15	0.10
Dichlorobenzenes (Sum)	238.6			0.06	44.01	60.70		0.68	0.27	0.16			5.45	0.27	0.6	0.3	0.21
Ethylbenzene	175.7				7.40												
Toluene	41.8	1.10			33.50	9.09		1.10	0.50	0.43	0.07		19.38		1	2.01	1.20
Trichlorobenzenes (Sum)	346.5					3.75											
Xylene (Total)	560.8				23.18												
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0																
Hexachlorobenzene	16.4			0.44		73.26		0.38					0.98	0.25	0.27	0.40	
Phenol	45.0																
Naphthalene	917.4	0.59		1.53	23.98	16.35			1.20	1.53	0.17		2.51	0.31			
Acenaphthene	860.7	0.74		1.03					0.64	0.99	0.36		1.63				
Fluorene	264.3	0.18	0.91	0.72		9.46			0.38	0.45			1.97				
Phenanthrene	542.7	0.36	1.44	0.92	2.58	2.03			0.83	0.92		0.995	3.69	0.33			
Anthracene	206.7	0.27	1.11	1.02		1.5			0.92	0.82			3.05	0.32			
Pyrene	343.8	1.57	0.38	4.07		4.4	2.18		3.20	3.78		3.20	9.6	1.08			
Benz(a)anthracene	191.5	0.97	3.34	2.56		3.0			2.04	2.30		2.66	7.3	0.89			
Chrysene	253.2	1.66	5.53	3.87		1.9			3.75	4.74	0.1	2.88	8.7	1.15			
Benzo(b)fluoranthene	908.4	0.34	2.42	1.76					1.98	2.97	0.13		3.19	0.58			
Benzo(a)pyrene	146.4	1.1	1.7	3.7					1.2	1.7	0.2	4.0		0.6			
Indeno(1,2,3-cd)pyrene	182.9	0.52	1.31	1.59					2.08	2.41			3.44				
Dibenz(a,h)anthracene	157.2	0.56		1.72		1.65							3.37	0.83			

Table 10-2. (cont.)

Chemical	Consensus PECs	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32
Benzo(g,h,i)perylene	779.7	0.60	**2.82**	**1.80**		**2.44**			**1.54**	**1.92**			**3.85**	**1.18**			
Acenaphthylene	1,300.7	0.33		**1.77**					**1.92**	0.85	0.28		**2.69**				
Benzo(k)fluoranthene	202.5	0.71		**1.63**		**2.02**			**1.43**	**2.22**	0.21		**2.9**	0.69			
Fluoranthene	1,436.3	0.43	**3.62**	**3.06**		0.91	0.51		0.77	0.97	0.45	0.8	**19.5**	0.38			
Pesticides/Polychlorinated Biphenyls																	
Chlordane (Sum)	5.1																
DDT and metabolites	29.6				**1.59**		0.41										
Aroclor-1016	110.9																
Aroclor-1248	203.7	0.27	**3.14**	**2.31**		**3.34**	**1.13**	0.32	0.39	0.42			**5.40**	0.49	0.29	0.46	0.24
Aroclor-1254	76.1											**1.31**					
Aroclor-1260	163.6		0.98	**1.47**		**1.83**		0.28					**6.11**			0.31	
PCBs (Sum)	294.8	0.30	**2.71**	**2.41**		**3.32**	0.95	0.38	0.51	0.49		0.59	**7.12**	0.48	0.32	0.49	0.30
Exceedances		5	17	23	14	23	12	8	17	16	0	17	29	10	8	8	8
Total Hazard Index		**7**	**42**	**49**	**155**	**228**	**27**	**15**	**34**	**36**	**0**	**37**	**326**	**26**	**12**	**14**	**13**

Table 10-2. (cont.)

Chemical	Consensus PECs	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48
Metals	**mg/kg**																
Antimony	3.6														0.91		
Arsenic	2.4				**4.69**										0.63		
Cadmium	2.4	**1.43**	0.88	0.46	0.40	0.27	0.5	**1.05**	0.97	**1.01**	0.93	0.93	**1.10**	0.97	0.42	0.63	0.42
Chromium	50.3	**1.26**	**1.62**	**3.60**	**13.71**	**3.87**	**4.19**	**3.97**	**1.57**	**1.29**	**1.30**	**1.26**	**1.36**	**1.03**	0.44	**2.22**	0.68
Copper	32.9	**2.05**	**1.07**	**1.44**	**4.22**	0.60	0.79	**1.77**	**1.63**	**1.55**	**1.65**	**1.71**	**2.01**	0.63	0.29	0.77	0.30
Lead	34.5	**2.21**	0.36	**1.79**	**7.27**	**1.12**	**1.6**	**1.22**	0.4	**1.6**	**1.7**	**1.58**	**2.14**	0.39	0.32	0.32	0.43
Manganese	278.3				**1.83**										0.95		
Mercury	2.2	**1.13**	0.07	0.45	0.59	0.22	0.32	0.77	**1.27**	**1.18**	**1.27**	**1.09**	**1.13**	0.30	0.18	0.22	0.21
Nickel	16.4	**1.61**	0.43	**2.38**	**13.13**	**2.06**	**2.85**	**3.57**	**1.77**	**1.77**	**1.70**	**1.58**	**1.47**	0.24	0.38	**2.46**	**1.03**
Selenium	0.6				0.72										0.62		
Silver	1.3				0.73										0.53		
Vanadium	5.6				**18.02**												
Zinc	88.2	**2.04**	0.62	**1.06**	**1.89**	0.48	0.66	**1.73**	**1.80**	**1.59**	**1.76**	**1.71**	**2.02**	0.49	0.35	0.90	0.42
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4	0.05						0.06	0.73	0.06	0.11						**5.59**
Chlorobenzene	428.4	0.04		0.04				0.03			0.09	0.08	0.08				
Dichlorobenzenes (Sum)	238.6	0.07			**44.03**	0.02		0.03	0.08		0.08	0.10	0.15				
Ethylbenzene	175.7																
Toluene	41.8	0.22	**5.02**	0.22	0.43		0.69	**2.39**	0.60	0.26	0.24	0.36		0.98			**2.13**
Trichlorobenzenes (Sum)	346.5																
Xylene (Total)	560.8																
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0																
Hexachlorobenzene	16.4			0.31						0.21	0.29	0.22	0.30				
Phenol	45.0																
Naphthalene	917.4			**1.74**		0.19	0.53	0.53	0.76					0.83		0.93	0.35
Acenaphthene	860.7		0.36				**1.51**		**2.0**							0.66	
Fluorene	264.3		0.16	**1.44**			**1.40**		0.6						0.16	0.37	
Phenanthrene	542.7		0.41	**1.64**			0.31	0.18	0.6					0.57	0.66	0.57	0.17
Anthracene	206.7		0.22	**1.98**			**1.0**	0.20	0.6					0.10	0.53	0.43	0.14
Pyrene	343.8		**1.08**	**5.24**		0.16	**1.2**	0.18	**1.9**					0.3	**1.69**	**1.92**	0.52
Benz(a)anthracene	191.5		0.57	**7.31**		0.32	**1.3**	0.43	**1.4**					0.8	**1.57**	**1.46**	0.31
Chrysene	253.2		0.83	**10.7**		0.55	**1.7**	0.75	**2.5**					0.1	**1.22**	**1.90**	0.43
Benzo(b)fluoranthene	908.4		0.63	**2.31**		0.17		0.15	0.6					0.23	0.48	0.29	0.07
Benzo(a)pyrene	146.4		0.3	**8.9**		0.4	**1.8**	0.4	**2.3**						**1.6**	**1.8**	0.3
Indeno(1,2,3-cd)pyrene	182.9			**4.21**					**3.44**					0.44	0.55	0.82	0.45
Dibenz(a,h)anthracene	157.2			**4.20**			0.89		**1.08**					0.21	0.27	0.53	

Table 10-2. (cont.)

Chemical	Consensus PECs	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48
Benzo(g,h,i)perylene	779.7		0.53	3.46		0.97	0.62	0.22	1.41					1.27	0.13	0.45	0.17
Acenaphthylene	1,300.7		0.29	2.31			0.55	1.31						0.35	0.03	0.32	
Benzo(k)fluoranthene	202.5		0.24	5.43		0.39	0.69		1.14					0.9		1.14	0.23
Fluoranthene	1,436.3		1.32	11.14	6.20	0.97	1.04	1.95	1.46					4.2	0.42	0.97	0.39
Pesticides/Polychlorinated Biphenyls																	
Chlordane (Sum)	5.1																
DDT and metabolites	29.6																
Aroclor-1016	110.9																
Aroclor-1248	203.7	0.31							0.45	0.35	0.59	0.69	0.74				
Aroclor-1254	76.1					0.85											
Aroclor-1260	163.6									0.24	0.44	0.45	0.47				
PCBs (Sum)	294.8	0.35				0.33			0.57	0.38	0.65	0.72	0.77				
Exceedances		7	5	20	10	3	11	9	15	7	6	6	7	3	4	7	3
Total Hazard Index		**12**	**10**	**82**	**115**	**7**	**20**	**19**	**27**	**10**	**9**	**9**	**11**	**6**	**6**	**13**	**9**

Table 10-2. (cont.)

Chemical	Consensus PECs	S49	S50	S51	S52	S53	S54	S55	S56	S57	S58	S59	S60	S61	S62	S63	S64
Metals	**mg/kg**																
Antimony	3.6																
Arsenic	2.4			**1.80**		**2.14**											
Cadmium	2.4	**1.14**	0.63	**1.18**	**1.05**	0.41	**1.7**	**5.98**	0.72	0.93	**1.08**	**1.39**	**1.26**	**1.31**	**1.01**	0.72	**1.05**
Chromium	50.3	**1.27**	**1.09**	**1.19**	**1.29**	0.35	**1.35**	**5.62**	**1.03**	**1.12**	**1.13**	**1.31**	**1.22**	**1.59**	0.86	0.75	1.00
Copper	32.9	**1.64**	**1.43**	**1.77**	**1.84**	0.36	0.69	**2.86**	**1.33**	**1.49**	**1.57**	**1.68**	**1.72**	**1.08**	0.41	**1.26**	**1.48**
Lead	34.5	**1.76**	**1.63**	**1.64**	**2.09**	0.32	0.2	**3.36**	**1.4**	**1.7**	**1.7**	**1.86**	**1.95**	0.51	0.27	**1.55**	**1.32**
Manganese	278.3			0.99		**1.15**											
Mercury	2.2	**1.36**	**1.54**	**1.13**	**1.27**	0.10	0.82	**5.87**	**1.45**	**1.63**	**1.25**	**1.13**	**1.09**	0.43	0.43	**1.45**	**1.13**
Nickel	16.4	**1.55**	**1.63**	**1.91**	**1.48**	0.66	**3.05**	**4.40**	**1.69**	**1.66**	**1.63**	**1.84**	**1.58**	0.56	**1.22**	**1.05**	**1.73**
Selenium	0.6			**1.62**													
Silver	1.3					0.94											
Vanadium	5.6			**2.07**		0.77											
Zinc	88.2	**1.80**	**1.66**	**1.73**	**2.00**	0.39	**1.28**	**3.06**	**1.58**	**1.70**	**1.69**	**1.77**	**1.85**	0.85	0.97	**1.51**	**1.53**
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4	0.14				0.74	**35.25**	**14.63**	0.05	0.14	0.07	0.09					0.07
Chlorobenzene	428.4	0.07		0.06							0.04	0.05					0.05
Dichlorobenzenes (Sum)	238.6							**3.31**			0.04						0.08
Ethylbenzene	175.7					0.08											
Toluene	41.8	0.26	0.26		0.26	0.16				0.36	0.38	0.41				1	0.26
Trichlorobenzenes (Sum)	346.5																
Xylene (Total)	560.8					0.02											
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0																
Hexachlorobenzene	16.4	0.37			0.24						0.23	0.23	0.29	0.73			
Phenol	45.0					**1.00**											
Naphthalene	917.4					0.37		**1.42**	0.45					0.65			
Acenaphthene	860.7													0.85			
Fluorene	264.3								0.3					0.45			
Phenanthrene	542.7			0.35			0.11	0.64	0.2					0.88	0.06		
Anthracene	206.7			0.42			0.1	0.48	0.3					0.63	0.08		
Pyrene	343.8			**1.13**		0.22	0.4	0.35	**1.2**					**2.5**	0.19		
Benz(a)anthracene	191.5			0.89		0.26	0.4	0.43	0.9					**1.8**	0.21		
Chrysene	253.2			0.99		0.23	0.4		**1.3**					**2.1**	0.28		
Benzo(b)fluoranthene	908.4							0.41	0.4					0.39	0.05		
Benzo(a)pyrene	146.4			**1.5**			0.6	**1.2**	**1.2**					**1.9**	0.3		
Indeno(1,2,3-cd)pyrene	182.9						0.32		**1.26**						0.18		
Dibenz(a,h)anthracene	157.2						0.19	0.58	0.63					0.62	0.13		

Table 10-2. (cont.)

Chemical	Consensus PECs	S49	S50	S51	S52	S53	S54	S55	S56	S57	S58	S59	S60	S61	S62	S63	S64
Benzo(g,h,i)perylene	779.7						0.21	0.68	0.67					0.67	0.19		
Acenaphthylene	1,300.7							1.54	0.52					0.59			
Benzo(k)fluoranthene	202.5						0.33	1.33	0.69					1.2	0.14		
Fluoranthene	1,436.3			0.20		0.05	0.11	6.06	0.97					0.7	0.13		
Pesticides/Polychlorinated Biphenyls	**µg/kg**																
Chlordane (Sum)	5.1																
DDT and metabolites	29.6																
Aroclor-1016	110.9																
Aroclor-1248	203.7	0.54	0.30		0.39			1.96		0.64	0.35	0.74	0.59	0.34		0.23	0.21
Aroclor-1254	76.1			1.01													
Aroclor-1260	163.6	0.36						0.73		0.35	0.24	0.43	0.36	0.32			
PCBs (Sum)	294.8	0.57	0.33	0.48	0.40			1.76		0.63	0.37	0.75	0.61	0.41		0.28	0.25
Exceedances		7	6	13	7	2	5	16	10	6	7	7	7	8	2	5	6
Total Hazard Index		**11**	**9**	**20**	**11**	**3**	**43**	**64**	**13**	**9**	**10**	**11**	**11**	**14**	**2**	**7**	**8**

Table 10-2. (cont.)

Chemical	Consensus PECs	S65	S66	S67	S68	S69	S70	S71	S72	S73	S74	S75	S76	S77	S78	S79	S80
Metals	**mg/kg**																
Antimony	3.6																
Arsenic	2.4		0.30														
Cadmium	2.4	**1.18**	0.55	0.38	**3.29**		**1.7**		**1.18**	0.10	0.33	0.37	0.59	0.72		0.46	**1.18**
Chromium	50.3	**1.04**	0.45	0.66	**2.24**	0.65	0.99	0.13	0.66	0.19	0.27	0.24	0.22	0.35	0.44	0.68	0.97
Copper	32.9	**1.51**	0.39	0.37	**1.35**	**1.21**	**1.47**	0.16	0.59	0.14	0.14	0.44	0.30	0.47	**1.10**	**1.18**	**1.39**
Lead	34.5	**1.47**	0.26	0.35	**1.21**	**1.32**	**1.5**	0.21	0.5	0.2	0.2	0.29	0.34	0.27	1.00	**1.17**	**1.42**
Manganese	278.3		0.65														
Mercury	2.2	1.00	0.10	0.42	**2.49**	**1.36**	0.86		0.36		0.38	**1.13**	0.86	**1.18**	**1.09**	**1.32**	0.95
Nickel	16.4	**1.54**	0.26	**1.11**	**1.97**	**1.15**	**1.29**	0.16	0.44	0.11	0.35	0.73	0.68	**1.24**	**1.06**	**1.39**	**1.53**
Selenium	0.6																
Silver	1.3		**2.11**														
Vanadium	5.6																
Zinc	88.2	**1.69**	0.44	0.57	**2.05**	**1.44**	**1.68**	0.27	0.90	0.19	0.25	0.74	0.53	0.86	**1.28**	**1.30**	**1.49**
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4					0.09	0.09									0.05	0.04
Chlorobenzene	428.4															0.03	0.03
Dichlorobenzenes (Sum)	238.6	0.10					0.10									0.1	0.05
Ethylbenzene	175.7																
Toluene	41.8	0.45		0.29		0.45	0.50	0.33	0.55	0.14	0.22				0.60	0.43	0.41
Trichlorobenzenes (Sum)	346.5																
Xylene (Total)	560.8																
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0		0.91														
Hexachlorobenzene	16.4											**1.71**	0.35	0.25	0.52		
Phenol	45.0																
Naphthalene	917.4		0.40				0.47		0.46								
Acenaphthene	860.7		0.53	0.51			0.36			0.20							
Fluorene	264.3		**1.82**	0.17					0.3			0.13		0.15			
Phenanthrene	542.7		**4.24**	0.17	0.17		0.24		0.4			0.17	0.059	0.18			
Anthracene	206.7		**3.63**	0.18	0.22		0.2		0.3			0.15	0.06	0.15			
Pyrene	343.8		**6.40**	0.67	0.64		0.8		**1.4**	0.14	0.11	0.7	0.29	0.7			
Benz(a)anthracene	191.5		**7.83**	0.57	0.63		0.5		0.9	0.08	0.21	0.5	0.19	0.5			
Chrysene	253.2		**4.74**	0.87	0.83		0.7	0.10	**1.2**	0.14		0.7	0.28	0.8			
Benzo(b)fluoranthene	908.4		**1.76**	0.29	0.19		0.3	0.07	0.6	0.09	0.21	0.14	0.07	0.13			
Benzo(a)pyrene	146.4		**8.9**	0.6	0.9		0.8	0.2	0.9	0.2		0.7	0.3	0.7			
Indeno(1,2,3-cd)pyrene	182.9		**2.95**	0.26	0.41							0.46	0.21	0.33			
Dibenz(a,h)anthracene	157.2		**1.15**		0.39		0.32		0.46			0.31	0.15	0.30			

Table 10-2. (cont.)

Chemical	Consensus PECs	S65	S66	S67	S68	S69	S70	S71	S72	S73	S74	S75	S76	S77	S78	S79	S80
Benzo(g,h,i)perylene	779.7		0.59	0.50	0.42		0.29		0.46	0.15		0.24	0.12	0.31			
Acenaphthylene	1,300.7		0.03				2.15										
Benzo(k)fluoranthene	202.5				0.36		0.49	0.10	0.64	0.11	0.06	0.43	0.19	0.4			
Fluoranthene	1,436.3		1.60	0.26	0.31		0.61	0.06	0.65	0.08		0.38	0.1	0.3			
Pesticides/Polychlorinated Biphenyls	**µg/kg**																
Chlordane (Sum)	5.1																
DDT and metabolites	29.6																
Aroclor-1016	110.9																
Aroclor-1248	203.7	0.24			0.26	0.23			0.17						0.23	0.17	0.17
Aroclor-1254	76.1																
Aroclor-1260	163.6																
PCBs (Sum)	294.8	0.30			0.30	0.28			0.23						0.28	0.22	0.23
Exceedances		6	12	1	7	5	6	0	3	0	0	2	0	2	4	5	5
Total Hazard Index		**8**	**47**	**1**	**15**	**6**	**10**	**0**	**4**	**0**	**0**	**3**	**0**	**2**	**5**	**6**	**7**

Table 10-2. (cont.)

Chemical	Consensus PECs	S81	S82	S83	S84	S85	S86	S87	S88	S89	S90	S91	S92	S93	S94	S95	S96
Metals	**mg/kg**																
Antimony	3.6	**1.77**															
Arsenic	2.4	**1.17**									**1.55**			0.30			
Cadmium	2.4		0.24		0.32	0.34	0.88	0.67		0.63	0.7	0.88	0.72	0.31	0.40	**2.44**	
Chromium	50.3	0.21	0.14	0.15	0.31	0.38	0.75	0.75	0.61	0.68	0.71	0.96	0.67	0.50	0.39	**1.50**	0.68
Copper	32.9	0.65	0.49	0.62	0.85	**1.07**	**1.33**	0.36	**1.11**	**1.23**	**1.24**	**1.39**	0.42	0.27	0.26	0.96	**1.13**
Lead	34.5	0.85	0.62	0.70	0.47	**1.28**	**1.21**	0.30	**1.29**	**1.61**	**1.3**	**2.16**	0.3	0.2	0.2	0.65	**1.33**
Manganese	278.3	**1.21**									0.86			0.93			
Mercury	2.2	0.59	0.50	0.77	**1.22**	0.86	**1.27**	0.45	**1.13**	**1.45**	**1.32**	**1.22**	0.24	0.18	0.44	**1.36**	**1.36**
Nickel	16.4	0.71	0.67	0.83	0.90	0.97	**1.37**	0.33	**1.36**	**1.44**	**1.33**	**1.56**	0.24	0.24	0.24	0.38	**1.23**
Selenium	0.6	**1.36**									**1.90**						
Silver	1.3													0.94			
Vanadium	5.6	**1.23**									**1.37**						
Zinc	88.2	0.70	0.57	0.68	0.98	**1.20**	**1.58**	0.39	**1.50**	**1.58**	**1.41**	**1.73**	0.46	0.28	0.27	**1.20**	**1.52**
Volatile Organic Compounds	**µg/kg**																
Benzene	150.4					0.07	0.22			0.12		0.11					
Chlorobenzene	428.4																
Dichlorobenzenes (Sum)	238.6																
Ethylbenzene	175.7																
Toluene	41.8					0.98	0.67	0.31	0.60	**1.32**		0.53	**5.74**	0.29	0.19	0.62	**1.51**
Trichlorobenzenes (Sum)	346.5																
Xylene (Total)	560.8																
Semivolatile Organic Compounds	**µg/kg**																
Dibenzofuran	372.0																
Hexachlorobenzene	16.4		0.67	**1.34**	**1.16**	0.22				0.24							
Phenol	45.0																
Naphthalene	917.4						0.50										
Acenaphthene	860.7				0.76		**1.98**										
Fluorene	264.3		0.21	0.27	0.28										0.17	0.18	
Phenanthrene	542.7	0.33	0.33	0.39	0.44		0.12				0.17		0.04			0.03	
Anthracene	206.7	0.17	0.19	0.28	0.31		0.19									0.09	
Pyrene	343.8	**1.11**	**1.16**	**1.63**	**2.18**		0.73				0.5				0.09	0.7	
Benz(a)anthracene	191.5	0.89	0.73	**1.04**	**1.20**		0.46								0.11	0.5	
Chrysene	253.2	0.91	**1.07**	**1.38**	**1.90**		0.79				0.5				0.16	0.8	
Benzo(b)fluoranthene	908.4	0.36	0.39	0.25	0.39		0.47								0.05	0.44	
Benzo(a)pyrene	146.4	**1.4**	0.6	**1.2**	**2.0**		0.5				0.8				0.2	0.4	
Indeno(1,2,3-cd)pyrene	182.9	0.60	0.31	0.93	0.77		**2.24**								0.09	0.33	
Dibenz(a,h)anthracene	157.2			0.60	0.89		0.34								0.12	0.34	

Table 10-2. (cont.)

Chemical	Consensus PECs	S81	S82	S83	S84	S85	S86	S87	S88	S89	S90	S91	S92	S93	S94	S95	S96
Benzo(g,h,i)perylene	779.7	0.14		0.69	0.45		0.56						0.28			0.50	
Acenaphthylene	1,300.7				0.48												
Benzo(k)fluoranthene	202.5			0.79	$\boxed{1.33}$		0.35								0.13	0.40	
Fluoranthene	1,436.3	0.28	0.44	0.97	0.91		0.54				0.10				0.07	0.54	
Pesticides/Polychlorinated Biphenyls	**µg/kg**																
Chlordane (Sum)	5.1																
DDT and metabolites	29.6																
Aroclor-1016	110.9										0.81						
Aroclor-1248	203.7								0.25	0.27		0.16					0.15
Aroclor-1254	76.1																
Aroclor-1260	163.6																
PCBs (Sum)	294.8								0.31	0.32	0.54	0.25					0.24
Exceedances		7	2	5	7	3	7	0	5	6	8	5	1	0	0	4	6
Total Hazard Index		**9**	**2**	**7**	**11**	**4**	**11**	**0**	**6**	**9**	**11**	**8**	**6**	**0**	**0**	**7**	**8**

Table 10-2. (cont.)

Chemical	Consensus PECs	S97	S98	S99	S100	S101	S102	S103	S104	S105	S106	S107	S108	S109	S110	S111	S112	S113	S114
Metals	**mg/kg**																		
Antimony	3.6																		
Arsenic	2.4				0.19													0.46	0.41
Cadmium	2.4	0.8	0.76	0.88	0.42	0.8	0.72	0.67	**2.11**	**1.9**	0.63	0.97	**2.19**	**1.10**	**1.69**	0.3	0.84	0.15	0.27
Chromium	50.3	0.84	0.95	0.89	0.54	0.54	0.87	0.84	**1.67**	**1.53**	0.75	0.83	**1.37**	**1.11**	**1.47**	0.24	0.71	0.33	0.49
Copper	32.9	**1.29**	**1.37**	**1.21**		0.35	**1.29**	**1.32**	0.84	0.84	**1.05**	**1.13**	**1.05**	0.77	0.94		0.52		0.39
Lead	34.5	**1.5**	**1.50**	**1.4**	0.1	0.3	**1.45**	**1.3**	0.5	0.6	**1.26**	**1.46**	0.66	0.32	0.20	0.6	0.29	0.5	0.9
Manganese	278.3				**1.60**													**1.02**	**1.24**
Mercury	2.2	**1.63**	**1.13**	1.00	0.16	0.39	0.86	**1.04**	0.63	0.73	0.82	0.42	**1.04**	0.73	1.00	**1.32**	0.50	0.08	0.29
Nickel	16.4				0.24			**1.43**	0.54	0.46	**1.42**	**1.14**	0.80	0.37	0.53	0.37	0.49	0.29	0.31
Selenium	0.6				0.67														
Silver	1.3				**1.01**														0.59
Vanadium	5.6																	0.32	
Zinc	88.2	**1.55**	**1.55**	**1.47**	0.36	0.52	**1.51**	**1.62**	0.95	0.87	**1.33**	**1.49**	**1.26**	0.59	0.84	0.43	0.57	0.44	0.55
Volatile Organic Compounds	**µg/kg**																		
Benzene	150.4																		
Chlorobenzene	428.4																		
Dichlorobenzenes (Sum)	238.6																		
Ethylbenzene	175.7																		
Toluene	41.8							0.26	**1.05**			0.45	0.24			0.12	0.19		
Trichlorobenzenes (Sum)	346.5																		
Xylene (Total)	560.8																		
Semivolatile Organic Compounds	**µg/kg**																		
Dibenzofuran	372.0																	0.05	
Hexachlorobenzene	16.4																		
Phenol	45.0				**1.00**														
Naphthalene	917.4																		
Acenaphthene	860.7							0.5	0.43	0.73				0.12		0.44	0.59	0.0	
Fluorene	264.3							0.2	0.18				**1.63**	0.22	0.31			0.3	
Phenanthrene	542.7							0.1	0.14				0.22		0.48	0.06		0.7	0.16
Anthracene	206.7							0.2	0.26				**1.40**		0.37			**1.1**	0.11
Pyrene	343.8							0.5	0.79				0.32		**1.25**	0.1		**2.2**	0.44
Benz(a)anthracene	191.5							0.3	0.63				0.21		0.63			**2.6**	0.43
Chrysene	253.2							0.5	**1.07**				0.27		0.67			**2.0**	0.34
Benzo(b)fluoranthene	908.4							0.3	0.24	0.0			0.10		0.22	0.1		0.6	
Benzo(a)pyrene	146.4							0.4	0.9	0.1			0.3		0.9			**3.3**	0.5
Indeno(1,2,3-cd)pyrene	182.9								**3.99**						0.45		**1.15**	0.82	
Dibenz(a,h)anthracene	157.2								0.52				0.18		0.45			0.39	

Table 10-2. (cont.)

Chemical	Consensus PECs	S97	S98	S99	S100	S101	S102	S103	S104	S105	S106	S107	S108	S109	S110	S111	S112	S113	S114
Benzo(g,h,i)perylene	779.7							0.45	0.42	0.27			0.14		0.37	0.42	0.03	0.17	0.04
Acenaphthylene	1,300.7																	0.06	
Benzo(k)fluoranthene	202.5							0.25	0.40	0.18						0.29			
Fluoranthene	1,436.3							0.35	0.39	0.41			0.13	0.02	0.41	0.97	0.19	0.42	0.12
Pesticides/Polychlorinated Biphenyls	**µg/kg**																		
Chlordane (Sum)	5.1																		
DDT and metabolites	29.6																		
Aroclor-1016	110.9																		
Aroclor-1248	203.7		0.19	0.17							0.27	0.36							
Aroclor-1254	76.1																		
Aroclor-1260	163.6																		
PCBs (Sum)	294.8		0.25	0.24							0.30	0.39							
Exceedances		4	4	3	2	0	3	5	5	2	4	4	7	2	3	1	1	6	1
Total Hazard Index		**6**	**6**	**4**	**3**	**0**	**4**	**7**	**10**	**3**	**5**	**5**	**10**	**2**	**4**	**1**	**1**	**12**	**1**

Notes:

COC	Chemical -	chemical of concern
PEC	Probable -	Probable effect concentration

[a] Consensus PECs could not be determined for heptachlor and heptachlor epoxide because the concentrations of those COCs were not found over a sufficiently large range.

[b] A ratio >1.0 indicates that the observed COC concentration exceeded the PEC for that chemical.

Table 10-3. Summary of Ratios to Consensus PECs for COC Concentrations in Surface Sediments in Onondaga Lake in 2000[a,b,c]

Chemical	Consensus PECs	S301	S302	S303	S304	S305	S306	S307	S308	S309	S310	S311	S312	S313	S314	S315
Metals	**mg/kg**															
Antimony	3.6			0.22					0.28	0.64	0.18		1.0		0.36	0.33
Arsenic	2.4	1.30	1.38	0.00	2.01	1.42	1.05	2.93	1.76	4.86	1.97	4.11	2.01	2.09	5.66	2.05
Cadmium	2.4	0.33	0.19	0.34	0.09	0.08	0.35	0.17	1.47	5.31	1.60	0.16	0.37	1.14	1.81	1.18
Chromium	50.3	0.58	0.49	0.65	0.37	0.48	0.35	7.93	9.93	14.62	3.04	0.53	0.93	1.62	2.80	1.66
Copper	32.9	1.14	1.25	1.43	1.16	1.09	0.36	2.07	3.49	5.92	2.57	1.16	1.18	2.02	3.22	3.80
Lead	34.5	2.05	1.15	1.43	1.32	0.83	0.22	3.65	3.22	13.6	2.87	0.70	3.10	6.46	10.5	3.16
Manganese	278.3	0.85	1.40	1.65	1.28	1.18	1.23	1.49	1.68	1.31	1.48	1.47	0.59	0.72	0.93	1.25
Mercury	2.2	1.72	1.36	1.45	1.00	1.13	0.31	0.19	1.90	11.61	5.44	7.26	15.06	3.99	18.9	4.35
Nickel	16.4	0.98	1.03	1.31	1.20	0.94	0.92	8.06	9.65	8.49	2.79	1.40	2.52	2.16	3.60	1.86
Selenium	0.6	1.10		3.62	2.07	1.90	2.07	2.76		2.07				2.07	1.55	
Silver	1.3	0.20		0.54					1.64	2.11	1.79		0.34	0.78	2.18	4.76
Vanadium	5.6	1.37	2.43	3.09	2.03	2.55	1.71	4.89	12.61	6.07	4.16	1.80	0.91	2.62	2.44	3.50
Zinc	88.2	1.46	1.38	1.88	1.01	1.27	0.78	0.65	2.72	3.00	2.13	0.76	0.90	2.15	2.11	2.73
Volatile Organic Compounds	**µg/kg**															
Benzene	150.4									20.6	0.43	5.32	61.19	0.53	17.3	0.07
Chlorobenzene	428.4							0.01	0.47	280	3.74	4.20	2334	44.4	257	0.33
Dichlorobenzenes (Sum)	238.6				0.34				0.54	260	3.81	1.93	704	41.1	65.8	1.22
Ethylbenzene	175.7									39.9	0.05	7.97	34.2	0.38	24.5	0.07
Toluene	41.8									114.8		88.5	136	2.39	13.6	0.23
Trichlorobenzenes (Sum)	346.5												15.6			
Xylene (Total)	560.8															
Semivolatile Organic Compounds	**µg/kg**															
Dibenzofuran	372.0							0.26						217.7	7.26	
Hexachlorobenzene	16.4	1.17	1.13	0.15	46.09	3.27	1.68		0.10	0.87	1.35	1.31	20.63		412.08	0.93
Phenol	45.0											48.89				
Naphthalene	917.4							0.06		76.30	1.42	21.80	45.78	78.5	50.14	
Acenaphthene	860.7							0.19						98.8		
Fluorene	264.3							0.49		6.81		1.93		530	14.00	
Phenanthrene	542.7		0.33	0.29	0.29			3.32	0.24	10.14		2.95		700	17.69	0.90
Anthracene	206.7							2.81	0.48			4.55		460		
Pyrene	343.8	0.20	1.19	0.99	0.84			9.31	0.87	11.35	1.63	1.25		436		2.65
Benz(a)anthracene	191.5		1.04	0.84	0.68			11.49	0.84	8.88	1.57			522	13.58	3.13
Chrysene	253.2		1.18	1.03	0.63			9.08	0.83	6.32	1.34			395	12.64	3.28
Benzo(b)fluoranthene	908.4		0.28	0.23	0.14			1.65	0.22		0.39			62.8	2.20	0.81
Benzo(a)pyrene	146.4		1.71	1.43	0.89			12.98	1.43		2.46			443.89	12.29	4.58
Indeno(1,2,3-cd)pyrene	182.9		1.20	0.98	0.50			6.56	0.87		1.15			208		2.79
Dibenz(a,h)anthracene	157.2							3.18						108		1.72

Table 10-3. (cont.)

Chemical	Consensus PECs	S301	S302	S303	S304	S305	S306	S307	S308	S309	S310	S311	S312	S313	S314	S315
Benzo(g,h,i)perylene	779.7		0.32	0.24	0.12			1.54	0.23		0.33			44.9		0.69
Acenaphthylene	1,300.7							0.05			0.18	0.29		**14.61**	**1.38**	0.15
Benzo(k)fluoranthene	202.5		**1.23**	0.94	0.59			**6.91**	0.89		**1.38**			**296**	**8.89**	**3.06**
Fluoranthene	1,436.3	0.06	0.33	0.25	0.19			**2.65**	0.22	**2.99**	0.38	0.52		**174**	**5.57**	0.91
Pesticides/Polychlorinated Biphenyls	**µg/kg**															
Chlordane (Sum)	5.1	0.29	0.33							0.80	**1.43**				**9.91**	0.23
DDT and metabolites	29.6	0.27	0.07		0.05	0.08		0.39	0.05	0.27	0.30	0.24	0.24	**2.98**	**2.00**	0.17
Aroclor-1248	203.7															
Aroclor-1254	76.1	**1.16**	0.32	0.33	0.14	0.25		**1.63**	0.64	**4.47**	**2.25**		0.92		**13.3**	0.84
Aroclor-1260	163.6	0.35	0.20	0.20		0.08			0.20	**2.71**			0.20	**6.24**	**4.19**	0.19
PCBs (Sum)	294.8	0.98	0.25	0.20	0.09	0.11		0.68	0.50	**11.0**	**2.30**		**6.29**	**10.9**	**37.4**	0.57
Exceedances		9	15	10	9	8	5	21	12	27	24	18	14	33	33	19
Total Hazard Index		12	20	18	58	14	8	107	52	936	56	208	3383	4925	1057	53

Table 10-3. (cont.)

Chemical	Consensus PECs	S316	S317	S318	S319	S320	S321	S322	S323	S324	S325	S326	S327	S328
Metals	mg/kg													
Antimony	3.6		0.22		0.14		0.08	0.24			0.11		0.61	
Arsenic	2.4	0.50	1.80	0.46	1.68	1.97	0.39	3.60	1.51	0.75	1.15	1.30	9.05	1.34
Cadmium	2.4	0.03	1.35	0.93	1.73	1.77	0.67	6.28	5.18	0.22	0.15	0.80	2.65	0.46
Chromium	50.3	0.13	1.64	0.47	1.40	1.70	0.52	2.66	2.94	0.75	3.93	8.48	83.04	7.07
Copper	32.9	0.92	3.58	1.39	2.79	3.62	0.92	4.65	2.96	0.41	0.93	2.16	10.45	1.32
Lead	34.5	1.46	4.11	1.52	3.30	3.36	0.81	8.26	2.65	0.33	0.85	2.26	8.78	1.43
Manganese	278.3	0.38	1.04	0.42	1.00	1.49	0.65	1.55	1.26	1.17	2.39	1.89	4.28	1.03
Mercury	2.2	0.16	7.80	0.41	1.72	2.77	0.10	0.45	0.73	0.15	0.07	0.59	1.04	0.33
Nickel	16.4	0.28	2.09	0.54	1.61	1.82	0.38	3.11	2.25	0.71	6.02	10.63	102.00	6.78
Selenium	0.6		2.76		2.42					2.42	3.26	3.97	10.18	
Silver	1.3	0.09	3.82	0.28	1.87	3.12	0.16	4.06	1.87			0.59	1.79	
Vanadium	5.6	0.96	2.89	0.73	2.71	4.12	0.66	5.25	1.73	1.00	3.96	10.15	56.92	4.62
Zinc	88.2	0.60	3.05	0.95	2.79	3.20	1.00	4.34	3.66	0.61	1.38	1.93	3.95	1.12
Volatile Organic Compounds	µg/kg													
Benzene	150.4		0.09	0.01		0.03		0.17		4.85	0.03	0.01	0.59	
Chlorobenzene	428.4		0.84	0.03	0.13	0.11		0.54	0.13	0.01		0.01	0.93	
Dichlorobenzenes (Sum)	238.6	0.71	4.88	0.75	0.84			0.88						
Ethylbenzene	175.7					0.03		1.88		0.06	0.00		0.10	
Toluene	41.8		0.12			0.08				3.35			0.50	
Trichlorobenzenes (Sum)	346.5													
Xylene (Total)	560.8									0.12	0.02	0.01	0.16	
Semivolatile Organic Compounds	µg/kg													
Dibenzofuran	372.0	0.46			0.19			1.37						
Hexachlorobenzene	16.4		1.65	0.49	1.29	0.89		0.21	0.21		0.13	0.11		0.07
Phenol	45.0													
Naphthalene	917.4	0.28		0.06	0.25			0.62			0.08	0.10		
Acenaphthene	860.7	0.51	0.19	0.08	0.20			2.56						
Fluorene	264.3		0.61	0.26	0.61									
Phenanthrene	542.7	2.76	2.03	0.81	2.40	0.50	0.44	14.37	0.41		0.16	0.29		0.14
Anthracene	206.7	4.84	5.32	1.02	3.00	0.68	0.53	12.10			0.03	0.29		
Pyrene	343.8	9.31	7.56	2.04	7.85	1.83	1.77	20.07	1.31	0.17	0.40	0.87		0.25
Benz(a)anthracene	191.5	9.40	9.92	1.83	8.36	1.83	1.88	13.06	1.31	0.21	0.40	0.94		0.30
Chrysene	253.2	7.11	8.69	1.50	6.71	1.78	1.46	12.24	1.34	0.18	0.39	0.87		0.21
Benzo(b)fluoranthene	908.4	1.32	1.87	0.28	1.32	0.40	0.35	1.87	0.31	0.04	0.10	0.21		0.05
Benzo(a)pyrene	146.4	13.66	14.34	2.53	11.61	3.07	2.94	15.71	1.71	0.33	1.02	1.37		0.33
Indeno(1,2,3-cd)pyrene	182.9	6.01	5.47	1.26	5.47	1.69	1.53	6.56	1.09		0.38	0.82		0.19
Dibenz(a,h)anthracene	157.2	2.86	2.74	0.59	2.23	0.83	0.61	3.31				0.27		

Table 10-3. (cont.)

Chemical	Consensus PECs	S316	S317	S318	S319	S320	S321	S322	S323	S324	S325	S326	S327	S328
Benzo(g,h,i)perylene	779.7	1.67	1.41	0.32	1.41	0.45	0.36	1.80	0.29		0.10	0.12		
Acenaphthylene	1,300.7	0.24	0.40	0.06	0.27		0.06	0.43						
Benzo(k)fluoranthene	202.5	**6.91**	**7.90**	**1.43**	**6.42**	**2.12**	**1.88**	**8.89**	**1.19**	0.20	0.36	0.89		0.26
Fluoranthene	1,436.3	**2.09**	**2.78**	0.53	**1.81**	0.44	0.34	**4.46**	0.33	0.04	0.12	0.27	0.70	0.07
Pesticides/Polychlorinated Biphenyls	**µg/kg**													
Chlordane (Sum)	5.1		0.29			**1.22**		**4.98**	**1.90**			0.45	**6.47**	
DDT and metabolites	29.6	0.05	0.27		0.23	0.34		**1.57**	0.60		0.01	0.28	0.41	0.07
Aroclor-1248	203.7									0.09	0.31	0.60		0.20
Aroclor-1254	76.1		**1.29**	**1.05**	**1.97**	**2.27**	0.19	**8.91**	**5.85**	0.21	0.82	**1.38**	**6.69**	0.66
Aroclor-1260	163.6		0.22	0.24	0.47	0.49	0.11	**1.38**	0.79	0.07	0.16	0.26	0.90	0.14
PCBs (Sum)	294.8		**1.02**	0.85	**1.59**	**1.97**	0.17	**6.17**	**6.05**	0.16	0.52	0.92	**5.72**	0.39
Exceedances		13	28	10	26	20	6	30	19	4	8	11	15	8
Total Hazard Index		69	115	16	87	47	11	187	48	12	23	46	313	25

Table 10-3. (cont.)

Chemical	Consensus PECs	S329	S330	S331	S332	S333	S334	S335	S336	S337	S338	S339	S340	S341
Metals	**mg/kg**													
Antimony	3.6	0.11	0.09		0.12	0.94	0.97	0.80	0.14	0.25	0.64	0.41	0.13	0.22
Arsenic	2.4		0.54	2.22	3.52	19.82	7.54	8.63	2.39	3.73	8.63	6.33	3.56	3.94
Cadmium	2.4	0.09	0.08	2.36	0.80	0.46	0.46	0.42	0.10	1.77	4.38	0.97	0.05	0.08
Chromium	50.3	0.93	1.66	6.40	4.41	4.29	16.15	10.69	3.77	20.66	62.18	47.08	0.25	0.20
Copper	32.9	0.15	0.25	3.04	2.71	5.74	7.20	5.89	1.30	5.04	10.85	11.12	1.28	1.12
Lead	34.5	0.13	0.38	3.36	2.31	2.60	11.39	5.79	2.19	4.98	12.92	17.76	0.62	0.90
Manganese	278.3	0.99	0.78	1.19	1.63	0.79	1.17	1.05	1.07	2.06	2.71	2.82	0.57	0.90
Mercury	2.2	0.05	0.20	0.86	1.36	0.73	1.90	0.95	3.49	6.98	5.49	2.90	1.13	1.13
Nickel	16.4	0.32	1.37	6.29	4.31	3.73	12.58	9.71	4.36	25.90	59.37	65.96	0.59	0.91
Selenium	0.6			2.24	2.76	5.87	6.56	5.87	1.55	4.66	6.56	5.18	1.50	2.76
Silver	1.3			2.57	0.31	0.28	0.37	0.41		1.87	2.89	0.77	0.38	0.34
Vanadium	5.6	0.37	0.95	6.76	2.21	5.94	9.87	11.33	7.71	23.73	43.36	29.98	1.11	2.30
Zinc	88.2	0.25	0.35	3.58	1.39	1.32	2.12	2.10	1.55	3.72	4.77	2.65	0.45	0.53
Volatile Organic Compounds	**µg/kg**													
Benzene	150.4		0.07		2.73	28.60	31.26	279.32	1.66	2.46	1.93	27.93	16.63	14.63
Chlorobenzene	428.4		0.35	0.11	6.54	1.00	4.44	210.10	20.78	151.74	44.36	112.05	8.40	2.33
Dichlorobenzenes (Sum)	238.6			0.21	25.57		1.97	1.84	25.82	1.76	2.47	9.64	62.88	15.93
Ethylbenzene	175.7				9.11	6.83	3.53	1.88				2.05	4.44	3.93
Toluene	41.8					13.16	16.27						26.32	95.71
Trichlorobenzenes (Sum)	346.5												1.44	0.95
Xylene (Total)	560.8			0.02	0.30	6.24	11.06	1.02				3.21	41.02	23.18
Semivolatile Organic Compounds	**µg/kg**													
Dibenzofuran	372.0				3.76	37.63	3.49	2.12			0.51	1.59	1.34	0.65
Hexachlorobenzene	16.4			0.12	4.27	0.24	0.30	2.62	0.07	0.12	0.66	0.16	8.73	1.47
Phenol	45.0													9.78
Naphthalene	917.4				41.42	141.7	6.00	16.35	0.39		0.36	8.39	19.62	17.44
Acenaphthene	860.7				1.16	8.71	1.16	0.52			0.26	0.57	0.48	0.20
Fluorene	264.3			0.22	6.43	68.1	9.46	5.30			1.25	3.33	3.40	1.59
Phenanthrene	542.7			0.18	4.61	77.4	18.43	7.37	1.33	0.61	2.40	8.48	8.29	3.69
Anthracene	206.7					77.4	19.35	8.71	0.68	1.06	3.15	7.74	4.02	2.52
Pyrene	343.8		0.09	0.73		43.6	27.93	9.89	2.56	1.45	3.49	11.1	6.40	3.20
Benz(a)anthracene	191.5			0.57	3.66	37.6	31.34	10.45	1.93	1.57	3.13	12.0	4.96	2.66
Chrysene	253.2			0.63	2.96	24.5	22.91	8.69	1.78	1.78	2.84	10.3	3.48	2.13
Benzo(b)fluoranthene	908.4			0.13		3.63	4.40	1.65	0.36	0.31	0.56	1.98	0.69	0.36
Benzo(a)pyrene	146.4			0.89		25.95	30.05	10.24	2.46	2.25	3.76	12.98	4.23	2.32
Indeno(1,2,3-cd)pyrene	182.9			0.48			15.31	5.19	1.31	1.09	1.80	6.56	1.97	1.15
Dibenz(a,h)anthracene	157.2						8.27	2.86			0.83	3.31		0.57

Table 10-3. (cont.)

Chemical	Consensus PECs	S329	S330	S331	S332	S333	S334	S335	S336	S337	S338	S339	S340	S341
Benzo(g,h,i)perylene	779.7			0.12			3.33	1.26	0.37		0.50	1.67	0.47	0.28
Acenaphthylene	1,300.7				1.38	3.15	**1.15**	0.59		0.09	0.15	0.36	0.30	0.18
Benzo(k)fluoranthene	202.5			0.54		**21.73**	**20.74**	**7.90**	**1.53**	**1.28**	**2.02**	**8.39**	**2.77**	**1.58**
Fluoranthene	1,436.3			0.17	**1.25**	**13.92**	**8.35**	**3.20**	0.68	0.53	**1.18**	**3.83**	**2.16**	**1.04**
Pesticides/Polychlorinated Biphenyls	**µg/kg**													
Chlordane (Sum)	5.1			0.98	0.23				0.25	0.63	0.49			
DDT and metabolites	29.6			0.34	0.05		0.25	0.13	0.12	0.16	**1.07**	0.72	0.14	
Aroclor-1248	203.7													
Aroclor-1254	76.1		0.15	**1.84**	0.32	0.68	**1.11**	**1.30**	0.83	**2.26**	**10.34**	**3.60**	0.83	0.29
Aroclor-1260	163.6			0.32	0.08		0.23	0.21	0.14	0.46	**1.61**	0.82	0.12	
PCBs (Sum)	294.8		0.12	**1.53**	0.78	0.35	0.61	0.73	0.36	**1.45**	**9.67**	**2.52**	0.78	**1.05**
Exceedances		0	2	13	24	27	34	31	20	24	29	32	25	25
Total Hazard Index		0	3	43	141	690	378	660	91	275	321	454	241	219

Table 10-3. (cont.)

Chemical	Consensus PECs	S342	S343	S344	S345	S346	S347	S348	S349	S350	S351	S352	S353	S354
Metals	**mg/kg**													
		0.22	0.16	**1.49**	0.19	0.13	0.15	0.16		0.16		0.33	0.13	0.33
Arsenic	2.4	**4.06**	**3.06**	**3.23**	**4.94**	**2.01**	**4.36**	**3.10**	**2.89**	**2.97**	**1.42**	**2.18**	**1.47**	**2.35**
Cadmium	2.4	0.59	0.37	0.42	0.15	0.38	0.26	0.36	0.59	0.30	0.35	**1.10**	**0.93**	**1.31**
Chromium	50.3	0.71	0.41	**1.32**	0.38	0.36	0.37	0.39	0.88	0.72	0.94	**1.73**	**1.21**	**2.24**
Copper	32.9	0.79	**4.01**	**2.70**	**1.54**	**1.77**	0.84	0.84	**1.52**	**1.25**	0.96	**2.63**	**2.16**	**2.83**
Lead	34.5	0.61	1.00	**8.89**	0.58	**1.05**	0.61	**2.66**	**1.46**	**2.26**	**2.57**	**21.73**	**2.93**	**2.67**
Manganese	278.3	**1.23**	0.75	**1.23**	**1.22**	0.83	**1.46**	**1.17**	0.88	**1.25**	0.83	0.67	0.63	**1.98**
Mercury	2.2	0.31	**1.27**	**35.24**	0.27	0.82	0.63	**1.41**	0.91	**4.58**	**2.77**	**8.25**	**5.31**	**1.50**
Nickel	16.4	**1.30**	**1.66**	**3.04**	**1.55**	**1.25**	**1.19**	**1.27**	**1.42**	**3.87**	**1.56**	**1.98**	**1.98**	**2.48**
Selenium	0.6		**2.07**				**2.59**		**1.90**		**2.07**	**1.35**		
Silver	1.3		0.30	0.66		0.23	0.12	0.22	0.32		0.48	**1.40**	0.40	**2.57**
Vanadium	5.6	**1.70**	**1.34**	**1.66**	**2.71**	**1.14**	**2.05**	**1.32**	**1.66**	**1.41**	**1.30**	**2.53**	**1.20**	**5.50**
Zinc	88.2	0.65	0.54	**1.47**	0.35	0.61	0.72	0.61	**1.13**	0.74	1.00	**2.47**	**1.84**	**2.81**
Volatile Organic Compounds	**µg/kg**													
Benzene	150.4	**1.20**	**7.98**		**9.98**	**5.05**	**2.66**		**10.64**	0.80	**1.26**	0.73	0.15	0.49
Chlorobenzene	428.4	0.54	**21.01**	**280.14**	**8.64**	**11.67**	**23.34**	**8.40**	0.51	**5.60**	**4.67**	**23.11**	0.49	0.11
Dichlorobenzenes (Sum)	238.6	**3.19**	**47.79**	**1001.82**	**12.20**	**9.72**	**209.59**	**34.37**	**2.56**	**21.80**	**48.67**	**40.24**	**12.78**	0.71
Ethylbenzene	175.7	**3.36**	**7.97**	**11.96**	**34.73**	**12.52**	**2.73**	**6.83**	**3.59**	**6.26**	0.07	**3.99**	0.02	0.02
Toluene	41.8	**11.01**	**52.64**		**198.59**	**57.42**	**22.73**	**16.99**	**98.10**	**9.09**	1.12	**3.83**		**1.12**
Trichlorobenzenes (Sum)	346.5			**101.02**			**10.68**			0.98	**6.93**			
Xylene (Total)	560.8	**17.65**	**60.63**	**13.91**	**267.50**	**94.52**	**17.48**	**28.53**	**16.94**	**24.97**	0.30	**2.85**	0.02	0.01
Semivolatile Organic Compounds	**µg/kg**													
Dibenzofuran	372.0									**1.91**	**4.57**	**20.96**	**1.85**	
Hexachlorobenzene	16.4	**1.62**	**2.16**			**3.23**	**2.28**		**5.65**	**12.88**	**9.10**	**11.84**	**7.14**	0.45
Phenol	45.0	**42.2**			**20.89**				**57.78**	**4.22**	**5.56**			
Naphthalene	917.4	**34.9**	**78.48**	**185.3**	**61.04**	**18.53**	**11.99**	**53.41**	**25.07**	**37.06**	**4.47**	**27.25**	**1.53**	0.34
Acenaphthene	860.7									0.27	**3.02**	**13.94**	**1.39**	
Fluorene	264.3		**4.54**			**1.44**			**2.08**	**2.84**	**6.81**	**45.40**	**4.54**	
Phenanthrene	542.7	**1.77**	**4.24**	**8.85**		**1.42**			**4.42**	**4.05**	**12.35**	**70.03**	**18.43**	0.68
Anthracene	206.7		**10.64**			**1.84**			**4.11**	**3.44**	**27.58**	**77.42**	**19.35**	**1.02**
Pyrene	343.8					**1.80**			**4.36**	**2.73**	**28.22**	**75.63**	**37.82**	**2.15**
Benz(a)anthracene	191.5								**2.25**	**1.83**	**25.07**	**50.14**	**47.01**	**2.04**
Chrysene	253.2								**2.09**	**1.62**	**20.54**	**39.50**	**35.55**	**2.21**
Benzo(b)fluoranthene	908.4									0.22	3.96	**7.38**	**7.60**	0.52
Benzo(a)pyrene	146.4									**1.37**	**30.05**	**55.32**	**64.19**	**3.07**
Indeno(1,2,3-cd)pyrene	182.9										**12.57**	**22.96**	**29.52**	**1.91**
Dibenz(a,h)anthracene	157.2										**5.03**	**8.91**	**15.27**	

Table 10-3. (cont.)

Chemical	Consensus PECs	S342	S343	S344	S345	S346	S347	S348	S349	S350	S351	S352	S353	S354
Benzo(g,h,i)perylene	779.7										3.33	5.77	8.34	0.54
Acenaphthylene	1,300.7	0.68			0.67				0.37	0.38	0.58	1.69	0.85	
Benzo(k)fluoranthene	202.5									**1.09**	**22.22**	**33.58**	**35.06**	**2.12**
Fluoranthene	1,436.3					0.43			0.97	0.97	**9.05**	**22.28**	**11.84**	0.58
Pesticides/Polychlorinated Biphenyls	**µg/kg**													
Chlordane (Sum)	5.1						0.21		0.54	0.27	0.84	**7.95**		0.50
DDT and metabolites	29.6	0.04		0.38		0.05	0.05		0.24	0.48	0.12	0.90	**1.32**	0.15
Aroclor-1248	203.7													
Aroclor-1254	76.1		0.41	**8.44**			0.26			**2.10**	**3.84**	**5.24**	**4.73**	**1.42**
Aroclor-1260	163.6	0.08		**3.40**	0.12	0.10	0.11	0.08	**1.46**	0.59	**2.30**	**1.01**	**1.27**	0.24
PCBs (Sum)	294.8	0.59	0.62	**39.7**	0.69	0.73	0.51	0.50	**1.42**	**1.71**	**4.13**	**3.49**	**3.21**	0.98
Exceedances		13	17	20	13	17	14	12	23	26	32	38	30	20
Total Hazard Index		125	311	1715	626	226	315	159	255	164	318	729	388	45

Table 10-3. (cont.)

Chemical	Consensus PECs	S355	S356	S357	S358	S360	S361	S362	S363	S364	S365	S366	S367	S368	S369	S370
Metals	**mg/kg**															
Antimony	3.6							0.09								
Arsenic	2.4	**2.51**		0.37		**1.23**	0.78	0.41	**1.11**	**2.14**	**3.81**	0.26	0.30	0.29	0.18	0.32
Cadmium	2.4	**1.39**	0.11	0.50		0.15	0.12		0.49	0.14	**1.14**	0.06	0.54	0.16	0.08	0.75
Chromium	50.3	**1.59**	0.18	0.38	0.06	0.38	0.22	0.08	0.73	0.41	**2.90**	0.21	0.83	0.47	0.15	**1.09**
Copper	32.9	**3.10**	0.18	0.35	0.05	0.74	0.36	0.12	0.60	0.55	**1.33**	0.18	0.62	0.36	0.13	0.56
Lead	34.5	**2.72**	0.12	0.16	0.02	0.81	0.75	0.08	0.46	0.39	**2.07**	0.1	0.28	0.21	0.09	0.36
Manganese	278.3	**1.83**	**1.17**	**1.02**	0.70	**1.16**	0.87	0.71	0.70	**1.42**	**1.71**	0.82	**1.04**	**1.14**	0.69	0.66
Mercury	2.2	**1.36**	0.07	0.27		**2.09**	0.16	0.04	0.12		0.29		0.17	0.10	0.02	0.25
Nickel	16.4	**2.01**	0.20	0.28	0.18	0.69	0.50	0.23	**1.06**	0.88	**3.49**	0.26	0.42	0.29	0.18	0.41
Selenium	0.6		**1.26**	**1.30**		0.15	0.89					**1.61**	**1.32**		**1.61**	
Silver	1.3	**2.73**		0.04					0.34		0.22		0.05			0.21
Vanadium	5.6	**4.76**	0.41	0.41	0.43	**1.66**	0.81	0.39	**1.08**	**1.59**	**2.84**	0.66	0.79	0.70	0.46	0.33
Zinc	88.2	**2.91**	0.38	0.45	0.12	**1.00**	0.60	0.20	**1.16**	0.44	**1.09**	0.31	0.69	0.38	0.25	0.63
Volatile Organic Compounds	**µg/kg**															
Benzene	150.4	0.13							**6.35**	0.05						
Chlorobenzene	428.4	0.10														
Dichlorobenzenes (Sum)	238.6	0.67														
Ethylbenzene	175.7	0.04							**5.57**							
Toluene	41.8	**2.15**							**170**							
Trichlorobenzenes (Sum)	346.5															
Xylene (Total)	560.8	0.01														
Semivolatile Organic Compounds	**µg/kg**															
Dibenzofuran	372.0															
Hexachlorobenzene	16.4	0.48				**2.86**			0.04		0.19					
Phenol	45.0					**11.27**			**1.5**	**26.67**						
Naphthalene	917.4	0.19							**10.9**		0.21					
Acenaphthene	860.7															
Fluorene	264.3															
Phenanthrene	542.7	**1.01**		0.10		0.02							0.06			
Anthracene	206.7	**1.31**		0.00									0.05			
Pyrene	343.8	**3.78**		0.43		0.38	0.19			0.38	0.29		0.35			0.20
Benz(a)anthracene	191.5	**3.55**		0.54		0.32	0.23						0.18			
Chrysene	253.2	**3.71**		0.38		0.37	0.18						0.14			
Benzo(b)fluoranthene	908.4	0.90		0.08		0.07							0.03			
Benzo(a)pyrene	146.4	**5.53**											**1.84**			0.40
Indeno(1,2,3-cd)pyrene	182.9	**3.17**				0.06							0.10			
Dibenz(a,h)anthracene	157.2	**1.40**											0.05			

Table 10-3. (cont.)

Chemical	Consensus PECs	S355	S356	S357	S358	S360	S361	S362	S363	S364	S365	S366	S367	S368	S369	S370
Benzo(g,h,i)perylene	779.7	0.90				0.07							0.03			
Acenaphthylene	1,300.7	0.12														
Benzo(k)fluoranthene	202.5	$\boxed{3.26}$		0.36		0.34							0.14			
Fluoranthene	1,436.3	0.97		0.11		0.11	0.04						0.08			0.05
Pesticides/Polychlorinated Biphenyls	**µg/kg**															
Chlordane (Sum)	5.1													0.35		
DDT and metabolites	29.6	0.18				0.07		0.06			0.12			0.10		0.04
Aroclor-1248	203.7															
Aroclor-1254	76.1	0.98				0.38			0.20		0.64		0.02			
Aroclor-1260	163.6	0.21				0.03		0.48	0.11		0.57		0.25	0.10		
PCBs (Sum)	294.8	0.71				0.70		0.36	0.17		0.48		0.23	0.13		
Exceedances		21	2	2	0	7	0	0	9	4	9	1	3	1	1	1
Total Hazard Index		56	2	2	0	21	0	0	199	32	20	2	4	1	2	1

Table 10-3. (cont.)

Chemical	Consensus PECs	S371	S359	S372	S373	S374	S400	S401	S402	S403	S404	S405	S406	S407	S434	S435
Metals	**mg/kg**															
Antimony	3.6			0.11	0.10	0.17										
Arsenic	2.4	0.59	**2.22**	0.36	0.81	0.31									0.26	**4.15**
Cadmium	2.4	0.93	0.03	**1.56**	**1.53**	0.26										0.10
Chromium	50.3	0.71	0.23	**1.62**	**1.05**	0.38									0.53	**3.16**
Copper	32.9	0.83	0.45	**1.04**	0.86	0.30									0.14	**1.72**
Lead	34.5	0.53	0.50	0.37	0.94	0.13									0.11	**1.12**
Manganese	278.3	0.94	**1.10**	**1.21**	0.77	0.97									0.54	0.56
Mercury	2.2	0.22	0.86	0.73	0.37	0.15	0.12	**2.74**	0.74	0.44	**11.28**	**5.44**	**27.92**	**4.24**	0.03	0.08
Nickel	16.4	0.58	0.77	0.46	0.82	0.29									0.56	**4.43**
Selenium	0.6		**1.28**	**1.66**	**2.20**	**2.76**									**2.07**	**9.32**
Silver	1.3	0.27		0.45	0.84											**1.33**
Vanadium	5.6	**1.59**	**2.19**	0.32	**1.30**	0.54									0.77	**4.07**
Zinc	88.2	**1.16**	0.59	0.95	**1.55**	0.47									0.28	0.57
Volatile Organic Compounds	**µg/kg**															
Benzene	150.4														0.03	**200**
Chlorobenzene	428.4														0.01	**6.77**
Dichlorobenzenes (Sum)	238.6															
Ethylbenzene	175.7														0.01	**404**
Toluene	41.8															**165**
Trichlorobenzenes (Sum)	346.5															
Xylene (Total)	560.8															**588**
Semivolatile Organic Compounds	**µg/kg**															
Dibenzofuran	372.0															
Hexachlorobenzene	16.4	0.24	0.61													
Phenol	45.0															
Naphthalene	917.4															**28,340**
Acenaphthene	860.7															
Fluorene	264.3															
Phenanthrene	542.7	0.55	0.83		0.14										0.13	**1,161**
Anthracene	206.7	0.43	0.45													
Pyrene	343.8	**1.57**	**1.40**		0.70										0.67	
Benz(a)anthracene	191.5	**1.51**	**1.20**		0.54										0.68	
Chrysene	253.2	**1.26**	**1.07**		0.51										0.59	
Benzo(b)fluoranthene	908.4	0.32	0.19		0.13										0.15	
Benzo(a)pyrene	146.4	**2.32**	**1.37**		0.46										0.75	
Indeno(1,2,3-cd)pyrene	182.9	**1.09**	0.55		0.45										0.53	
Dibenz(a,h)anthracene	157.2															

Table 10-3. (cont.)

Chemical	Consensus PECs	S371	S359	S372	S373	S374	S400	S401	S402	S403	S404	S405	S406	S407	S434	S435
Benzo(g,h,i)perylene	779.7	0.28	0.14		0.11										0.14	
Acenaphthylene	1,300.7															
Benzo(k)fluoranthene	202.5	**1.58**	0.84		0.56										0.64	
Fluoranthene	1,436.3	0.44	0.42		0.17										0.12	
Pesticides/Polychlorinated Biphenyls	**µg/kg**															
Chlordane (Sum)	5.1															
DDT and metabolites	29.6		0.04	0.04												0.10
Aroclor-1248	203.7															
Aroclor-1254	76.1	0.26	0.49		0.31		0.16	**5.19**		**2.24**		**6.03**	**2.17**	**2.07**	0.17	
Aroclor-1260	163.6	0.12			0.10		0.02	**1.00**		0.08		**1.03**	0.79	0.71		0.10
PCBs (Sum)	294.8	0.13	0.22		0.22		0.08	**4.33**	**1.14**	**3.86**	**8.73**	**4.28**	**9.91**	**1.61**	0.12	0.45
Exceedances		8	8	5	5	1	0	4	1	2	2	4	3	3	1	15
Total Hazard Index		12	12	7	8	3	0	13	1	6	20	17	40	8	2	30,895

[a] Consensus PECs could not be determined for heptachlor and heptachlor epoxide.

[b] A ratio >1.0 indicates that the observed COC concentration exceeded the PEC for that chemical. Exceedances are bolded.

[c] Sediments at Stations 400 to 407 (benthic chemistry locations) were analyzed for only mercury, methylmercury, and PCBs.

Table 10-4. Hazard Quotients for Measured Fish Concentrations

COC	Bluegill 95%UCL HQ NOAEL	Bluegill 95%UCL HQ LOAEL	Bluegill Mean HQ NOAEL	Bluegill Mean HQ LOAEL	Gizzard Shad 95%UCL HQ NOAEL	Gizzard Shad 95%UCL HQ LOAEL	Gizzard Shad Mean HQ NOAEL	Gizzard Shad Mean HQ LOAEL
Antimony	0**	0**	0**	0**	0*	0*	0*	0*
Arsenic	1.4	0.5	0.7	0.3	0*	0*	0*	0*
Chromium	61	18	16	4.6	0*	0*	0*	0*
Mercury	5.4	1.8	2.7	0.9	0*	0*	0*	0*
Methylmercury	3.5	1.2	2.8	0.9	2.3	0.8	2.1	0.7
Selenium	15	1.5	9.2	0.9	0*	0*	0*	0*
Vanadium	29	2.9	20	2.0	0*	0*	0*	0*
Zinc	3.2	2.7	2.1	1.8	0*	0*	0*	0*
Endrin	0.2	2.3E-02	0.1	1.5E-02	0*	0*	0*	0*
DDT and metabolites	4.7E-02	9.7E-03	3.9E-02	8.0E-03	0*	0*	0*	0*
Polychlorinated biphenyls	0.5	0.1	0.3	0.1	0*	0*	0*	0*
Dioxin/furan TEQ (Fish)	0.4	0.2	0.1	0.1	0*	0*	0*	0*

Table 10-4. (cont.)

CoC	Carp 95%UCL HQ NOAEL	Carp 95%UCL HQ LOAEL	Carp Mean HQ NOAEL	Carp Mean HQ LOAEL	Catfish 95%UCL HQ NOAEL	Catfish 95%UCL HQ LOAEL	Catfish Mean HQ NOAEL	Catfish Mean HQ LOAEL
Antimony	0**	0**	0**	0**	0.4	0.2	6.3E-02	3.5E-02
Arsenic	4.0	1.5	1.7	0.6	0**	0**	0**	0**
Chromium	21	6.2	7.2	2.1	5.7	1.7	3.1	0.9
Mercury	4.3	1.4	3.5	1.2	6.3	2.1	4.9	1.6
Methylmercury	4.8	1.6	3.9	1.3	7.8	2.6	7.1	2.4
Selenium	20	2.0	10	1.0	13	1.3	7.6	0.8
Vanadium	24	2.4	13	1.3	27	2.7	20	2.0
Zinc	13	11	6.1	5.2	2.2	1.8	1.2	1.0
Endrin	1.0	0.1	0.5	0.0	0.8	0.1	0.5	0.0
DDT and metabolites	0.4	0.1	0.3	0.1	0.6	0.1	0.3	0.1
Polychlorinated biphenyls	2.5	0.5	1.6	0.3	2.1	0.4	1.5	0.3
Dioxin/furan TEQ (Fish)	2.6	1.2	1.0	0.5	0.6	0.3	0.4	0.2

Table 10-4. (cont.)

CoC	White Perch 95%UCL HQ NOAEL	White Perch 95%UCL HQ LOAEL	White Perch Mean HQ NOAEL	White Perch Mean HQ LOAEL	SMB 95%UCL HQ NOAEL	SMB 95%UCL HQ LOAEL	SMB Mean HQ (NOAEL)	SMB Mean HQ (LOAEL)
Antimony	0.4	0.2	0.4	0.2	0**	0**	0**	0**
Arsenic	0**	0**	0**	0**	3.6	1.4	2.4	0.9
Chromium	2.5	0.7	2.5	0.7	3.2	0.9	2.3	0.7
Mercury	7.7	2.6	7.0	2.3	7.3	2.4	7.0	2.3
Methylmercury	12	4.1	11	3.6	8.2	2.7	7.2	2.4
Selenium	7.8	0.8	7.8	0.8	10	1.0	4.8	0.5
Vanadium	0**	0**	0**	0**	20	2.0	11	1.1
Zinc	0.5	0.4	0.5	0.4	1.6	1.4	1.1	0.9
Endrin	0.1	1.4E-02	0.1	1.2E-02	0.2	1.7E-02	0.2	1.6E-02
DDT and metabolites	0.2	3.5E-02	0.1	1.3E-02	0.1	2.1E-02	0.1	1.5E-02
Polychlorinated biphenyls	1.3	0.3	1.1	0.2	1.0	0.2	0.9	0.2
Dioxin/furan TEQ (Fish)	0.5	0.3	0.4	0.2	0.5	0.2	0.3	0.1

Table 10-4. (cont.)

CoC	LMB 95%UCL HQ NOAEL	LMB 95%UCL HQ LOAEL	LMB Mean HQ NOAEL	LMB Mean HQ LOAEL	Walleye 95%UCL HQ NOAEL	Walleye 95%UCL HQ LOAEL	Walleye Mean HQ NOAEL	Walleye Mean HQ LOAEL
Antimony	NA	NA	NA	NA	0**	0**	0**	0**
Arsenic	0*	0*	0*	0*	0**	0**	0**	0**
Chromium	0*	0*	0*	0*	**3.2**	0.9	**3.2**	0.9
Mercury	**6.9**	**2.3**	**6.6**	**2.2**	**15**	**5.2**	**14**	**4.6**
Methylmercury	0*	0*	0*	0*	**18**	**6.1**	**15**	**5.1**
Selenium	0*	0*	0*	0*	0**	0**	0**	0**
Vanadium	0*	0*	0*	0*	0**	0**	0**	0**
Zinc	0*	0*	0*	0*	0**	0**	0**	0**
Endrin	0**	0**	0**	0**	0.3	2.7E-02	0.1	1.3E-02
DDT and metabolites	0.1	1.2E-02	2.9E-02	6.1E-03	0.2	3.6E-02	0.1	2.1E-02
Polychlorinated biphenyls	0.7	0.1	0.4	0.1	**2.8**	0.6	**1.5**	0.3
Dioxin/furan TEQ (Fish)	**1.4**	0.7	0.9	0.4	0*	0*	0*	0*

Notes:

* denotes not analyzed

** denotes all non-detects

Hazard quotients equal to or greater than one are outlined and bolded.

DDT – dichlorodiphenyltrichloroethane

LMB – largemouth bass

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

HQ – hazard quotient

SMB – smallmouth bass

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 10-5. Hazard Quotients for Modeled Tree Swallow Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Metals				
Arsenic	0.1	4.4E-02	0.1	3.1E-02
Barium	**10**	**5.1**	**8.3**	**4.1**
Cadmium	**7.0**	0.5	**4.6**	0.3
Chromium	**53**	**11**	**57**	**11**
Copper	0.8	0.6	0.6	0.5
Lead	**1.8**	0.2	**1.3**	0.1
Methylmercury	**19**	**1.9**	**11**	**1.1**
Mercury	**6.5**	**3.3**	**3.1**	**1.5**
Nickel	0.2	0.1	0.2	0.1
Selenium	**6.8**	**3.4**	**5.4**	**2.7**
Thallium	NA	NA	NA	NA
Vanadium	0.1	1.1E-02	0.1	7.9E-03
Zinc	**6.4**	0.7	**5.6**	0.6
Volatile Organic Compounds				
Xylenes	NA	NA	NA	NA
Dichlorobenzenes	**3.0**	0.3	**1.4**	0.1
Trichlorobenzenes	NA	NA	NA	NA
Semivolatile Organic Compounds				
Bis(2-ethylhexyl)phthalate	0.7	0.1	0.6	0.1
Total polycyclic aromatic hydrocarbons	**287**	**29**	**292**	**29**
Pesticides/Polychlorinated Biphenyls				
DDT and metabolites	0.8	0.1	0.6	0.1
Total polychlorinated biphenyls	**1.9**	0.2	**1.8**	0.2
Dioxins/Furans				
Dioxins/furans (TEQ) avian	**5.6**	0.6	**1.3**	0.1

Note: Hazard quotients equal to or greater than one are outlined and bolded.

DDT – dichlorodiphenyltrichloroethane

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

HQ – hazard quotient

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 10-6. Hazard Quotients for Modeled Mallard Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Metals				
Barium	**2.4**	**1.2**	**1.8**	0.9
Cadmium	**1.0**	0.1	0.7	4.7E-02
Chromium	**10**	**2.1**	**9.7**	**1.9**
Copper	0.2	0.1	0.1	0.1
Methylmercury	**4.3**	0.4	**2.7**	0.3
Mercury	0.9	0.4	0.7	0.3
Nickel	3.9E-02	2.8E-02	3.7E-02	2.7E-02
Vanadium	2.6E-02	2.6E-03	1.5E-02	1.5E-03
Zinc	**1.2**	0.1	**1.0**	0.1
Volatile Organic Compounds				
Xylenes	NA	NA	NA	NA
Dichlorobenzenes	**2.1**	0.2	0.3	3.3E-02
Trichlorobenzenes	NA	NA	NA	NA
Organic Compounds				
Total polycyclic aromatic hydrocarbons	**393**	**39**	**118**	**12**
Pesticides/Polychlorinated Biphenyls				
Total polychlorinated biphenyls	0.4	3.9E-02	0.3	3.0E-02
Dioxins/Furans				
Dioxins/furans (TEQ) avian	**1.4**	0.1	0.3	3.1E-02

Note:

Hazard quotients equal to or greater than one are outlined and bolded.
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-7. Hazard Quotients for Modeled Belted Kingfisher Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Chromium	0.2	3.8E-02	0.2	3.6E-02
Lead	0.1	1.4E-02	0.1	8.7E-03
Methylmercury	**23**	**2.3**	**20**	**2.0**
Mercury	0.7	0.3	0.6	0.3
Selenium	3.9E-03	2.0E-03	3.1E-03	1.5E-03
Zinc	1.0E-02	1.1E-03	8.6E-03	9.5E-04
Organic Compounds				
Total polycyclic aromatic hydrocarbons	**12**	**1.2**	**3.7**	0.4
Pesticides/Polychlorinated Biphenyls				
Endrin	2.9E-04	2.9E-05	2.4E-04	2.4E-05
Hexachlorocyclohexanes	2.2E-05	7.2E-06	2.0E-05	6.3E-06
DDT and metabolites	**19**	**1.9**	**12**	**1.2**
Total polychlorinated biphenyls	**11**	**1.1**	**3.1**	0.3
Dioxins/Furans				
Dioxins/furans (TEQ) avian	**1.8**	0.2	**1.4**	0.1

Note:

Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-8. Hazard Quotients for Modeled Great Blue Heron Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Chromium	0.1	2.7E-02	0.1	2.5E-02
Methylmercury	**18**	**1.8**	**15**	**1.5**
Mercury	0.3	0.1	0.3	0.1
Selenium	0.5	0.2	0.4	0.2
Zinc	**1.1**	0.1	0.8	0.1
Organic Compounds				
Total polycyclic aromatic hydrocarbons	**4.0**	0.4	**1.2**	0.1
Pesticides/Polychlorinated Biphenyls				
Hexachlorocyclohexanes	1.0E-02	3.3E-03	0.2	0.1
DDT and metabolites	**8.0**	0.8	**5.3**	0.5
Total polychlorinated biphenyls	**2.7**	0.3	**1.4**	0.1

Note:

Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
UCL – upper confidence limit

Table 10-9. **Hazard Quotients for Modeled Osprey Exposure**

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Chromium	0.1	2.1E-02	0.1	1.9E-02
Methylmercury	**24**	**2.4**	**20**	**2.0**
Mercury	0.3	0.2	0.3	0.2
Selenium	0.7	0.4	0.5	0.3
Zinc	**1.6**	0.2	**1.2**	0.1
Pesticides/Polychlorinated Biphenyls				
Hexachlorocyclohexanes	1.5E-02	4.8E-03	0.3	0.1
DDT and metabolites	**9.3**	0.9	**6.3**	0.6
Total polychlorinated biphenyls	**2.5**	0.3	0.2	2.5E-02
Dioxins/Furans				
Dioxins/furans (TEQ) avian	0.6	5.6E-02	0.4	4.3E-02

Note:

Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-10. **Hazard Quotients for Modeled Red-Tailed Hawk Exposure**

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Chromium	0.2	4.7E-02	0.2	3.4E-02
Lead	0.4	4.2E-02	0.3	3.0E-02
Methylmercury	0.3	2.7E-02	7.2.E-02	7.2E-03
Mercury	0.1	7.1E-02	2.6.E-02	1.3E-02
Organic Compounds				
Total polycyclic aromatic hydrocarbons	**252**	**25**	**14**	**1.4**
Pesticides				
DDT and metabolites	**1.5**	0.2	0.3	3.3E-02
Dioxins/Furans				
Dioxins/furans (TEQ) avian	**9.9**	0.99	**1.01**	0.1

Notes:

1. All state wetlands (i.e., SYW- 6, 10, 12, and 19) surrounding Onondaga Lake and the dredge spoils area were included in soil pathway calculations.
2. Hazard quotients equal to or greater than one are outlined and bolded.

DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-11. Hazard Quotients for Modeled Little Brown Bat Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Antimony	0.2	2.0E-02	0.2	1.6E-02
Arsenic	**1.1**	0.1	0.8	0.1
Barium	**2.1**	**1.3**	**1.7**	**1.0**
Cadmium	**4.5**	0.5	**3.0**	0.3
Chromium	**7.2**	**1.8**	**7.8**	**1.9**
Cobalt	0.4	3.9E-02	0.3	3.4E-02
Copper	**1.4**	**1.1**	**1.1**	0.9
Lead	0.1	1.2E-02	0.1	8.8E-03
Manganese	3.8E-02	1.2E-02	3.5E-02	1.1E-02
Methylmercury	**21**	**2.1**	**13**	**1.3**
Mercury	**1.3**	0.1	0.6	0.1
Nickel	0.1	0.1	0.2	0.1
Selenium	0.2	0.1	0.2	0.1
Thallium	0.1	7.9E-03	0.1	7.1E-03
Vanadium	**2.7**	0.3	**1.9**	0.2
Zinc	0.3	0.1	0.2	0.1
Volatile Organic Compounds				
Trichlorobenzenes	2.8E-02	7.8E-03	0.1	1.7E-02
Xylenes	**2.3**	**1.9**	0.5	0.4
Semivolatile Organic Compounds				
Hexachlorobenzene	**6.0**	0.6	**4.6**	0.5
Total polycyclic aromatic hydrocarbons	**18**	**1.8**	**19**	**1.9**
Pesticides/Polychlorinated Biphenyls				
Dieldrin	0.6	0.3	0.5	0.2
Total polychlorinated biphenyls	0.4	0.1	0.4	0.1
Dioxins/Furans				
Dioxins/furans (TEQ) mammalian	**11**	**1.1**	**2.9**	0.3

Note:

Hazard quotients equal to or greater than one are outlined and bolded.

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

HQ – hazard quotient

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 10-12. Hazard Quotients for Modeled Short-Tailed Shrew Exposure

COC	SYW-6 95%UCL Hazard Quotient NOAEL	SYW-6 95%UCL Hazard Quotient LOAEL	SYW-6 Mean Hazard Quotient NOAEL	SYW-6 Mean Hazard Quotient LOAEL	SYW-10 95%UCL Hazard Quotient NOAEL	SYW-10 95%UCL Hazard Quotient LOAEL	SYW-10 Mean Hazard Quotient NOAEL	SYW-10 Mean Hazard Quotient LOAEL
Total Metals								
Antimony	0.4	3.6E-02	0.1	9.5E-03	8.3E-02	8.3E-03	4.5E-02	4.5E-03
Arsenic	2.0	0.2	1.1	0.1	5.3	0.5	2.3	0.2
Barium	0.1	0.1	0.1	0.1	0.1	7.3E-02	0.1	4.9E-02
Cadmium	11	1.1	3.5	0.4	1.2	0.1	0.7	0.1
Chromium	1.0	0.2	0.3	0.1	0.3	0.1	0.2	4.3E-02
Lead	1.5	0.1	0.7	0.1	1.0	0.1	0.6	0.1
Methylmercury	22	2.2	19	1.9	22	2.2	20	2.0
Mercury	0.2	1.9E-02	0.1	1.1E-02	0.2	1.7E-02	0.1	1.3E-02
Selenium	1.7	1.0	0.6	0.4	1.3	0.8	0.7	0.4
Thallium	2.6	0.3	1.4	0.1	4.3	0.4	2.8	0.3
Vanadium	2.9	0.3	1.8	0.2	3.9	0.4	2.0	0.2
Zinc	0.7	0.4	0.5	0.2	0.4	0.2	0.4	0.2
Volatile Organic Compounds								
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06	5.8E-06	1.6E-06	5.6E-06	1.6E-06
Semivolatile Organic Compounds								
Hexachlorobenzene	ND	ND	ND	ND	2.0	0.2	1.5	0.1
Total polycyclic aromatic hydrocarbons	213	21	47	4.7	155	15.5	38	3.8
Pesticides/Polychlorinated Biphenyls								
Chlordane	ND	ND	ND	ND	ND	ND	ND	ND
Dieldrin	ND	ND	ND	ND	ND	ND	ND	ND
Total polychlorinated biphenyls	3.9E-02	9.7E-03	2.8E-02	6.9E-03	0.1	3.5E-02	5.9E-02	1.5E-02
Dioxins/Furans								
Dioxins/furans (TEQ)	15	1.5	5.9	0.6	4.4	0.4	3.6	0.4

Table 10-12. (cont.)

COC	SYW-12 95%UCL Hazard Quotient NOAEL	SYW-12 95%UCL Hazard Quotient LOAEL	SYW-12 Mean Hazard Quotient NOAEL	SYW-12 Mean Hazard Quotient LOAEL	SYW-19 95%UCL Hazard Quotient NOAEL	SYW-19 95%UCL Hazard Quotient LOAEL	SYW-19 Mean Hazard Quotient NOAEL	SYW-19 Mean Hazard Quotient LOAEL
Total Metals								
Antimony	0.1	9.5E-03	4.7E-02	4.7E-03	0.2	1.8E-02	0.1	1.0E-02
Arsenic	1.4	0.1	0.99	9.9E-02	2.8	0.3	2.3	0.2
Barium	0.1	0.1	0.1	4.6E-02	0.3	0.2	0.2	0.1
Cadmium	7.5	0.8	5.0	0.5	2.5	0.3	1.6	0.2
Chromium	0.7	0.2	0.4	0.1	0.3	0.1	0.3	0.1
Lead	1.0	0.1	0.7	0.1	2.1	0.2	1.0	0.1
Methylmercury	19	1.9	19	1.9	29	2.9	27	2.7
Mercury	0.1	1.2E-02	9.4E-02	9.4E-03	0.6	6.3E-02	0.4	4.1E-02
Selenium	0.7	0.5	0.4	0.3	1.2	0.8	1.1	0.7
Thallium	ND	ND	ND	ND	ND	ND	ND	ND
Vanadium	2.0	0.2	1.1	0.1	1.7	0.2	1.6	0.2
Zinc	0.5	0.3	0.5	0.2	0.4	0.2	0.4	0.2
Volatile Organic Compounds								
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06	3.4	0.9	1.2	0.3
Semivolatile Organic Compounds								
Hexachlorobenzene	1.8	0.2	0.5	4.9E-02	783	78	241	24
Total polycyclic aromatic hydrocarbons	191	19	61	6.1	2,565	256	794	79
Pesticides/Polychlorinated Biphenyls								
Chlordane	0.1	2.6E-02	0.1	1.3E-02	0.6	0.1	0.2	4.2E-02
Dieldrin	1.1	0.6	0.6	0.3	7.3	3.7	5.0	2.5
Total polychlorinated biphenyls	0.4	0.1	0.2	0.1	1.8	0.5	1.4	0.4
Dioxins/Furans								
Dioxins/furans (TEQ)	NA	NA	NA	NA	1,706	171	681	68

Table 10-12. (cont.)

COC	Dredge Spoils 95%UCL HQ NOAEL	Dredge Spoils 95%UCL HQ LOAEL	Dredge Spoils Mean HQ NOAEL	Dredge Spoils Mean HQ LOAEL
Total Metals				
Antimony	0.1	6.5E-03	4.9E-02	4.9E-03
Arsenic	**2.7**	0.3	**1.9**	0.2
Barium	6.0E-02	3.6E-02	5.6E-02	3.3E-02
Cadmium	1.7E-04	1.7E-05	1.7E-04	1.7E-05
Chromium	0.2	4.6E-02	0.1	2.7E-02
Lead	0.2	1.7E-02	0.1	1.4E-02
Methylmercury	0.1	6.8E-03	5.E-02	5.E-03
Mercury	0.2	1.8E-02	9.E-02	9.E-03
Selenium	**1.1**	0.7	0.8	0.5
Thallium	ND	ND	ND	ND
Vanadium	**3.7**	0.4	**2.4**	0.2
Zinc	0.3	0.2	0.3	0.1
Volatile Organic Compounds				
Trichlorobenzenes	5.8E-06	1.6E-06	5.6E-06	1.6E-06
Semivolatile Organic Compounds				
Hexachlorobenzene	**38**	**3.8**	**4.6**	0.5
Total polycyclic aromatic hydrocarbons	**9.0**	0.9	**2.0**	0.2
Pesticides/Polychlorinated Biphenyls				
Chlordane	NA	NA	NA	NA
Dieldrin	NA	NA	NA	NA
Total polychlorinated biphenyls	3.4E-02	8.6E-03	1.7E-02	4.3E-03
Dioxins/Furans				
Dioxins/furans (TEQ)	0.7	0.1	0.4	4.2E-02

Notes:

NA = Not available
ND = Not detected
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level

Hazard quotients equal to or greater than one are outlined and bolded.
HQ – hazard quotient
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-13. Hazard Quotients for Modeled Mink Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Arsenic	0.2	1.7E-02	0.1	1.1E-02
Chromium	0.7	0.2	0.6	0.2
Methylmercury	**12**	**1.2**	**9.4**	0.9
Mercury	0.1	1.4E-02	0.1	9.9E-03
Selenium	0.1	8.2E-02	0.1	7.1E-02
Vanadium	0.3	2.8E-02	0.7	6.7E-02
Organic Compounds				
Hexachlorobenzene	**9.2**	0.9	**1.1**	0.1
Total polycyclic aromatic hydrocarbons	**33**	**3.3**	**4.5**	0.4
Pesticides/Polychlorinated Biphenyls				
DDT and metabolites	1.5E-02	2.9E-03	7.5E-03	1.5E-03
Dieldrin	0.2	0.1	0.1	0.1
Total polychlorinated biphenyls	**109**	**11**	**34**	**3.4**
Dioxins/Furans				
Dioxins/furans (TEQ) mammalian	**42**	**4.2**	**4.9**	**0.5**

Notes:

1. All State wetlands surrounding Onondaga Lake and the dredge spoils area were included in soil pathway calculations.
2. Hazard quotients equal to or greater than one are outlined and bolded.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 10-14. Hazard Quotients for Modeled River Otter Exposure

COC	95%UCL HQ NOAEL	95%UCL HQ LOAEL	Mean HQ NOAEL	Mean HQ LOAEL
Total Metals				
Arsenic	0.8	0.1	0.5	0.1
Chromium	0.3	0.1	0.3	0.1
Methylmercury	**43**	**4.3**	**36**	**3.6**
Mercury	0.1	1.5E-02	0.1	1.4E-02
Selenium	0.9	0.5	0.7	0.4
Vanadium	0.8	0.1	0.6	0.1
Organic Compounds				
Total polycyclic aromatic hydrocarbons	**5.2**	0.5	**1.6**	0.2
Pesticides/Polychlorinated Biphenyls				
DDT and metabolites	**5.9**	**1.2**	**2.3**	4.5E-01
Total polychlorinated biphenyls	**130**	**13**	**69**	**6.9**
Dioxins/Furans				
Dioxins/furans (TEQ) mammalian	**2.8**	0.3	**1.5**	0.2

Notes:

Hazard quotients equal to or greater than one are outlined and bolded.
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

11. UNCERTAINTY ANALYSIS (ERAGS STEP 7)

There are several sources of uncertainties associated with ecological risk estimates, and uncertainties are present at the various Ecological Risk Assessment Guidance for Superfund (ERAGS) steps, as discussed in this chapter. Sources of uncertainty in this BERA include:

- Sampling representativeness and analysis and quantitation error.
- Onondaga Lake conditions.
- Chemical of concern (COC)/stressor of concern (SOC) selection.
- Background and reference concentrations.
- Sediment effect concentrations (SECs) and macroinvertebrate analyses.
- The conceptual model.
- Natural variation and parameter error.
- Food-web model error.
- Toxicological studies used as measures of effect.

The following sections identify the strengths and limitations of the various components of this assessment.

11.1 Sampling Representativeness and Analysis and Quantitation Error

This section discusses the potential impact of sampling representativeness (locations and frequency) and analysis and quantitation error on the uncertainty inherent in the BERA.

11.1.1 Representativeness of Sampling Locations

The representativeness of the sampling locations selected for chemical analyses of surface water, surficial sediment, sediment porewater, surface soils, benthic macroinvertebrates, fish, and other media are discussed in this section.

11.1.1.1 Surface Water Sampling

The 1992 Remedial Investigation (RI) sampling program conducted by Honeywell/PTI analyzed lake water from the centers of the southern and northern basins and water from near the mouths of each tributary from April to November. At each of the two lake stations, samples were collected at 1-meter (m) intervals from the water surface to the lake bottom. Based on findings from the Onondaga County Department of Water Environment Protection (OCDWEP) monitoring program, it is known that the centers of the two basins are generally representative of water quality conditions throughout most of the deep portions of the lake due to lake circulation patterns and wind-induced mixing processes. The 1-m vertical sampling interval is commonly used in limnological studies (including the OCDWEP monitoring program for Onondaga Lake) and is considered sufficiently small to capture all major limnological features of the water column in Onondaga Lake. The sampling locations near the mouth of each tributary are considered representative because they were generally located upstream of the influence of the lake, and there were no major sources of inflow between each sampling station and the lake.

A limited amount of supplemental water column sampling was also conducted by Honeywell in 1999 to evaluate conditions during fall turnover at stations in the centers of both basins of the lake and to evaluate water quality from a human health perspective at nine nearshore stations. The 1999 data were not intended to provide a representative overview of the lake. The combined surface water database provides sufficient coverage of the lake, and little uncertainty is associated with sampling locations.

11.1.1.2 Surface Sediment Sampling

For the RI sampling, Honeywell/PTI evaluated surface sediment in 1992 at 114 stations distributed throughout the lake, with the densest coverage in areas near major known sources (i.e., between Tributary 5A and Ley Creek and at the mouths of other tributaries). Nineteen samples were fully characterized and the remaining 95 were partially characterized (see Chapter 7, Table 7-1 for details).

In 2000, Honeywell/PTI sampled an additional 84 stations in areas where the 1992 data indicated that additional information was needed (see Chapter 7, Table 7-2). Given the comprehensive and stratified nature of the station-location scheme used for the RI, most potential areas of concern are likely to have been sampled. The selection of stations to evaluate areas of concern may bias any Onondaga Lake-wide mean values for individual COCs or SOCs.

The surface sediments analyzed from the 1992 sampling were collected at a depth of 0 to 2 cm, while the 2000 surface sediment samples were collected at a depth of 0 to 15 cm. The variation in sampling location and depth adds uncertainty to comparisons between sampling years. Sediment samples from 0 to 2 cm do not accurately characterize the entire horizon of sediment that ecological receptors are exposed to (i.e., the biologically active zone). The overall affect of sampling from only 0 to 2 cm rather than the 0 to 15 cm is unknown. However, because the concentration of contaminants in Onondaga Lake sediments tends to increase with depth, it is possible that results from sampling 0 to 2 cm could underestimate the overall exposure of ecological receptors to contaminants.

11.1.1.3 Sediment Porewater Sampling

Sediment cores for porewater were collected at four locations in 1992 and seven locations in 2000. The chloride profiles from the 1992 porewater samples obtained in August and November are strikingly different throughout the length of the cores. The August concentrations are consistently and substantially lower, which could indicate a rapid mechanism for movement through the sediments; however, it most likely indicates that the lake water and porewater were allowed to mix during collection of the August samples, which would invalidate the samples. Because of the significant change in chloride concentrations, the August and possibly all of the porewater results from 1992 are suspect and unusable and, therefore, were not used in this analysis.

Porewater samples were also collected for analysis of mercury and methylmercury from three depths at seven locations in the lake in 2000 (see Chapter 8, Section 8.1.2.10). These data were considered acceptable for use in the RI and BERA.

11.1.1.4 Wetland Soils/Sediment and Surface Soil Sampling

Four wetland soil/sediment samples (i.e., 0 to 15 cm) were collected from each of four wetland areas around Onondaga Lake (i.e., New York State-regulated Wetlands SYW-6, SYW-10, SYW-12, and SYW-19) in 2000. Although these samples cover only a small portion of the wetland areas, they are considered adequate for a general characterization of wetlands in the context of the overall BERA. Because of the high mercury levels in 2000 at one station in Wetland SYW-6 (Station S375), NYSDEC sampled five additional locations in this wetland area in May 2002 to determine if the mercury was indicative of widespread contamination from the lake. These supplemental wetland SYW-6 data are also used in this BERA. Wetland SYW-10 will be further evaluated as part of the RI/FS for the Geddes Brook/Ninemile Creek site and Wetland SYW-19 will be further evaluated as part of the RI/FS for the Wastebed B/Harbor Brook site.

Eight surface soil samples were collected in the dredge spoils area in 2000 to characterize the quality of the fill material that exists in varying thicknesses over the more contaminated dredge spoils material. The dredge spoils samples did not have a depth distribution corresponding to depth intervals preferred for use in ecological assessments (i.e., 0 to 15 cm). As a result, the depth intervals used for assessing ecological exposure to surface soils included data from depths ranging from the surface to 15 to 107 cm. Most of the surface samples were collected from the relatively uniform cover material above the more-contaminated spoils; thus, no significant bias is expected from using the thicker surface soil profile and the surface soil data are deemed suitable for assessing ecological exposure. The dredge spoils area will be further evaluated as part of a separate site with its own investigation.

11.1.1.5 Benthic Macroinvertebrate Sampling

Benthic macroinvertebrates were analyzed for total and methylmercury in 1992 and 2000. The combined data set and its coverage of the lake are considered adequate for a general estimate of mercury concentrations in Onondaga Lake macroinvertebrates. Six macroinvertebrate samples were analyzed for polychlorinated biphenyls (PCBs) on an Aroclor basis in 1992. Macroinvertebrates were sampled from areas of the lake with the highest concentrations of contamination and are not representative of overall lake conditions, particularly if uptake rates vary depending on PCB concentration. Due to the small sample size, unrepresentative sampling, and analytical uncertainty, data from the US Army Corps of Engineers (USACE) database (2002) were used to derive a biota-sediment accumulation factor (BSAF) for PCBs. No other contaminants were analyzed in macroinvertebrates from the lake. Consequently, BSAFs were required to estimate most COC concentrations (see Section 11.8.1).

Sediment toxicity to macroinvertebrates was evaluated in 1992 and 2000. In 1992, sediment toxicity was analyzed at 79 stations in Onondaga Lake and five stations in Otisco Lake (which was used as the reference location), while in 2000 sediment toxicity was tested at 15 stations in Onondaga Lake and two stations in Otisco Lake. Comparison of the results from the 1992 macroinvertebrate toxicity testing locations to the 1992 sediment sampling locations indicates that most potential areas of concern were sampled. Therefore, these tests are considered to be representative of lake bottom conditions.

The purpose of the 2000 sediment toxicity testing was to evaluate potential differences in toxicity through the use of chronic test protocols versus the 1992 short-term protocols. Based on the results of the 2000 toxicity tests and associated sediment chemistry as compared to the 1992 test results, these analyses met their intended goals.

11.1.1.6 Fish Sampling

Hundreds of fish were analyzed by Honeywell and NYSDEC between 1992 and 2000 (Appendix I, Tables I-6, I-13, and I-15). The coverage of combined fish collected from the lake is considered good, but fish data sets may not be adequate to evaluate species-specific uptake or movement of COCs through the food web.

However, few fish samples were analyzed for the full suite of volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs) (including polycyclic aromatic hydrocarbons [PAHs]), and Target Analyte List (TAL) metals. Estimates of some of these contaminants may not accurately reflect concentrations of contaminants found in the lake. There is no systematic bias in the direction of these estimates. The larger fish data sets for mercury, pesticides, and PCBs are considered to accurately represent fish-flesh concentrations under current lake conditions.

11.1.1.7 Sampling of Other Media

A limited number of samples of other media (phytoplankton, zooplankton, and muskrat [*Ondatra zibethicus*]) were sampled (Chapter 7, Tables 7-1 and 7-2). In general, data from other media were inadequate to be considered representative of the lake. The mercury analyses of phytoplankton were used to estimate macrophyte intake in the mallard duck (*Anas platyrhynchos*) food model. There was some uncertainty associated with these data because concentrations were provided on a per-sample basis, rather than as parts per billion (ppb) (or another unit). Despite the uncertainty associated with these data they were used due to the absence of other plant data and a reliable uptake equation.

Representativeness of sampling of other media was not evaluated because of other issues that precluded data use. Wildlife receptors selected in this BERA do not feed directly on zooplankton (with the possible exception of the mallard duck, for which phytoplankton data were used) and, therefore, those data were not used in food-web modeling. Zooplankton data were also not needed to model fish concentrations, as fish body burdens were measured directly. As discussed in Chapter 8, Section 8.2.6.5, the muskrat data were not used because of questions related to the reliability of the transfer factors and the inappropriate use of a herbivorous mammal to represent small mammals, inclusive of insectivores, based on differences in bioaccumulation related to feeding strategies.

11.1.2 Representativeness of Sampling Frequency

For the 1992 RI sampling, both lake and tributary water were analyzed monthly between April and December 1992, when the lake was free from ice. In addition, the monthly sampling events for the tributaries included separate sampling events for different flow conditions. The monthly sampling frequency is considered sufficient for characterizing chronic exposure conditions in the lake and tributaries. The sampling of variable flow regimes in the tributaries allowed a broader range of water quality conditions to be assessed.

Other media sampling occurred primarily during the summer (see Chapter 7, Table 7-1), which is the period of greatest biological activity and considered an appropriate sampling time for BERA data. The 1999 sampling took place from September to December and the 2000 sampling occurred from July to September (Chapter 7, Table 7-2). Although this sampling covered a narrower time frame, the sampling frequency is considered to be acceptable as the data were collected primarily to fill in data gaps.

11.1.3 Analysis and Quantitation Uncertainties

The analysis and quantitation of analytical parameters was minimized by following quality assurance/quality control (QA/QC) protocols and using USEPA Contract Laboratory Program (CLP) laboratories. Data were validated prior to being entered in the Onondaga Lake database.

The exceptions to these protocols were field variables (i.e., pH, specific conductivity, temperature, and dissolved oxygen [DO]) that were determined in the field, and geotechnical analyses, such as grain size and density and solids, which were performed by a geotechnical laboratory.

All data provided by Honeywell were used with the exception of the 1992 PCB biota data. As part of NYSDEC's rewrite of the Honeywell-generated risk assessments, the quality of the historical data, including the 1992 data set, was revisited. As a result of this reevaluation of the fish data, NYSDEC decided to exclude the 1992 Honeywell/PTI PCB data set because it may have underestimated the actual total PCB concentration in Onondaga Lake fish fillets. This determination was based on a review of the Onondaga Lake RI/FS Bioaccumulation Investigation Data Report (PTI,1993b) prepared by PTI for Honeywell. The report describes the sampling and analysis methods for the 1992 fish collection and includes the analytical data and the quality assurance reviews.

The report indicates that PCBs were not detected in 973 of 1,232 (79 percent) of the samples. Of those samples in which PCBs were not detected, 793 (82 percent) were qualified for various factors of non-compliance with data quality objectives, including the possibility of false negatives. Surrogate recovery was poor, averaging around 43 percent. NYSDEC's data quality review indicated that the Honeywell 1992 data set consistently underestimated fish PCB concentrations, based on the laboratory's failure to meet data quality objectives (specifically, poor surrogate spike recovery) and the high percentage of samples that were reported as not detected for PCBs.

Despite the fact that a large amount of earlier (i.e., 1992) Honeywell PCB data were considered unusable, there are almost 200 NYSDEC (1992 to 2000) and Honeywell (2000) PCB analyses used in this BERA, covering both low molecular weight (either Aroclor 1016 or Aroclor 1242) and high molecular weight (reported as Aroclor 1254/1260) PCBs for fish tissue, which is considered sufficient for the BERA.

11.1.3.1 Mercury Methylation in Wetlands

Owing to the absence of site-specific data, mercury methylation in wetlands was estimated to be 1 percent, based on data from the LCP Bridge Street site (NYSDEC/TAMS, 1998a) and the literature (see Chapter 6, Section 6.3.1). This is considered to be a realistic estimate and resulted in risks up to an order-of-magnitude greater than one to the short-tailed shrew in all wetland areas. Even if mercury methylation rates are lower, for example the 0.25 percent average seen at the LCP Bridge Street site (Chapter 6, Table 6-3), the NOAEL for methylmercury would still be exceeded at all four wetland areas, indicating that methylmercury in wetlands poses a risk to ecological receptors. If methylmercury comprises more than 1 percent of the mercury found in wetland soils, hazard quotients (HQs) would be even higher.

11.1.4 Summary of Sampling and Analysis Uncertainties

The 1992 and 1999/2000 RI sampling programs provided sufficient sediment, soil, water, and fish data for overall coverage of the site, which includes the lake and Wetlands SYW-6 and SYW-12. Honeywell data were supplemented with NYSDEC fish data, NYSDEC Wetland SYW-6 data, and Onondaga County stressor data. All data were considered acceptable for use, with the exception of Honeywell's 1992 PCB fish data. The level of uncertainty associated with surface water, sediment, and fish mercury sampling frequency and locations is considered low. The data sets for other media were not as extensive, providing a moderate level of uncertainty, but were considered acceptable for use in the BERA.

11.2 Onondaga Lake Conditions – Uncertainties

There are a number of uncertainties associated with conditions other than COCs (e.g., stressor concentrations, effects of natural disturbances, etc.) in Onondaga Lake. For example, stressors may impact exposure or exclusion of biota from exposure (e.g., due to low levels of DO), or result in increased eutrophication, affecting the composition of ecological communities. Effects on organisms that provide food or habitat at the bottom of the food chain can affect upper trophic level receptors. In addition, stress on receptors may affect the toxicity of COCs. This section discusses major uncertainties in the Onondaga Lake ecosystem, such as macrophyte distribution and eutrophication-related stressors.

11.2.1 Factors Limiting the Distribution and Abundance of Macrophytes

Factors limiting the distribution and abundance of aquatic plants include the following:

- Light (availability, transparency, and depth).
- Water chemistry.

- Sediment chemistry.
- Sediment texture and composition.
- Temperature.
- Competition.
- Abiotic disturbance (e.g., wave action, water level).
- Biotic disturbance (e.g., herbivory).
- Stressors such as salinity (Madsen and Owens, 2000).

All these factors influence the species of macrophytes found in Onondaga Lake and their coverage. As discussed below, wave disturbance is an additional, significant factor for aquatic plant life.

Wave Disturbance

Sediment disturbance by waves can be a major limiting factor for macrophytes. However, the presence of macrophytes can reduce the action of waves. Therefore, the influence of wave disturbance can be greater in a damaged aquatic system than in a healthy unstressed system. Madsen et al. (1993) indicated that the combination of low transparency and moderate fluctuations of water level limits plant colonization in Onondaga Lake. Even though the level of water in Onondaga Lake is regulated by the dam (on the Oswego River at Phoenix, New York, downstream from the lake), variations in lake level can affect macrophyte communities. However, the rarity of floating-leaved aquatic plants forming defined wetlands along the shoreline of Onondaga Lake indicates that other factors (e.g., salinity, substrate, nutrient loadings, reduced water transparency) besides wave disturbance are also affecting plant growth and establishment.

Macrophytes and oncolites may exhibit a negative relationship with respect to their field distributions in Onondaga Lake. This negative relationship may be due to the direct or indirect effects of the oncolites on the macrophytes or an independent variable, such as wave-induced sediment disturbance. Wave disturbance can cause direct harm to the plants or it can have an adverse effect in combination with the coarse, nutrient-poor sediments found in Onondaga Lake. Laboratory studies showed macrophyte growth to be lower on all Onondaga Lake sediments than on reference sediments (Madsen et al., 1993).

The dominant macrophyte occurring at 391 of the 3,497 Onondaga Lake quadrats surveyed in 1991 was *Potamogeton pectinatus* (sago pondweed) (Madsen et al., 1993). *P. pectinatus* is a pioneering species and quickly inhabits newly flooded areas and invades shallow waters with relatively strong wave action or those that are polluted (Kantrud, 1990). It often shows mass development in areas where the environment became temporarily unsuitable for other species and belongs to a group of plants tolerant of, and able to maintain dominance in, altered ecosystems. These plants are also able to withstand rapid and considerable fluctuations in the salt content of the waters they inhabit and are able to tolerate a wide range of nutrient (nitrogen, phosphorus) concentrations (Kantrud, 1990). The small size of *P. pectinatus* beds indicates that this species and other macrophytes have recently reestablished themselves in the lake or extended their range due to water quality improvements in Onondaga Lake since the late 1980s (e.g., reduction of the high levels of salinity).

The recent reestablishment of macrophytes in the lake indicates that, although wave action may be a factor affecting the species and abundance of aquatic macrophytes found in Onondaga Lake, other stressors (e.g., reduced water transparency, salinity, oncolites, and calcium carbonate deposition) and COCs discussed in Chapter 8, Section 8.1 may be more significant in influencing the Onondaga Lake macrophyte community.

11.2.2 Effects of Calcium and Oncolites on the Macroinvertebrate Community

The effects of calcium and oncolites on benthic macroinvertebrate communities in Onondaga Lake are evaluated in this section. As noted previously, there is a strong association between the calcium carbonate content of sediments and the density of oncolites in the nearshore zone of Onondaga Lake. To evaluate whether the calcium carbonate content of sediments or oncolites are limiting to benthic macroinvertebrate communities in Onondaga Lake, correlation analyses were conducted using the Spearman rank correlation coefficient.

The species richness of benthic macroinvertebrate communities at depths up to 4.5 m exhibited positive significant correlations with the calcium carbonate content of sediment ($P \leq 0.05$) (Figure 11-1). Chironomid abundance showed no significant correlation ($P > 0.05$) with the calcium carbonate content of sediments, while amphipod abundance exhibited a significant correlation (Figure 11-2). These results indicate that with the increase of calcium carbonate content of sediment in Onondaga Lake there is a corresponding increase in taxa richness and amphipod abundance. However, there is no evidence linking this relationship directly to levels of calcium carbonate in the sediments, levels of contamination, or other physical properties of the sediments.

With respect to oncolites, the species richness of benthic communities and amphipod abundance both correlated significantly ($P \leq 0.05$) with the oncolite volume of the sediment at the depths evaluated (Figures 11-3 and 11-4). However, the amount of scatter in these plots makes any direct quantitative prediction relatively uncertain. Chironomid abundance was also correlated significantly ($P \leq 0.05$) with oncolite volume at the 1.5 m depth, but was not correlated significantly ($P > 0.05$) at the 4.5 m depth (Figure 11-4). These results indicate that with the increase of oncolites in Onondaga Lake sediment there is a corresponding increase in taxa richness and amphipod abundances. It is unknown whether this relationship is directly due to the levels of calcium carbonate in the sediments, the levels of contamination, or other physical properties of the sediments, such as providing increased numbers of microhabitats and refuges from predation.

Based on these uncertainties, it is difficult to define the potential direct effects of oncolites on the benthic community in Onondaga Lake. However, the benthic community should be considered in conjunction with the macrophyte community. If the presence of oncolites reduces macrophyte coverage, the onocolites themselves may provide an alternative microhabitat for benthic invertebrates to use. The oncolite habitat is likely to be of lower quality than the macrophyte habitat because of its lower productivity and complexity.

11.2.3 Oxic Hypolimnion

An evaluation was made of the future conditions that may occur in Onondaga Lake if its eutrophic conditions improve to the point that anoxic conditions in the hypolimnion are eliminated. Such changes would result in a series of chemical and biological responses in both the water column and sediment.

Oxic conditions could reduce the concentrations of some substances, such as ammonia, in the water column. The concentrations of at least some metals, including mercury, iron, and manganese, would decrease in the water column, as was evident during fall turnover. Currently there is a large production of methylmercury in the water column in the anoxic hypolimnion, which would be greatly reduced if the hypolimnion remained oxic year-round and there was an increase in the concentration of DO, both of which would make it possible for biota to inhabit the hypolimnion and profundal sediments throughout the year. However, depending on the oxygen demand of the sediments, oxygenating the water column could also merely shift the location of some oxidation reduction boundaries from the water column to the sediment, leaving contaminants available at the sediment-water interface.

The current absence of oxygen and the presence of acutely toxic substances precludes higher (eukaryotic) life and the direct uptake of contaminants in the hypolimnion, although there is some evidence that pelagic biota are presently exposed to the elevated methylmercury concentrations across the thermocline. However, if the decrease in acutely toxic contaminants and increased oxygen leads to the reestablishment of aquatic life in the hypolimnion and surface sediments, then those biota would have the opportunity to interact more extensively with the profundal sediments. These interactions would include direct contact with contaminated surface sediments, and would involve the potential for more deeply buried sediments with higher levels of contaminants to be brought to the surface by bioturbation. Mean concentrations of some bioaccumulative contaminants, including mercury, are currently found at higher levels in deep sediments than in shallow sediments (see Appendix I, Tables I-3 and I-4). This inventory of contaminants, particularly mercury, could be taken up by these organisms and introduced into the food chain. The absence of life in the hypolimnion currently precludes uptake of these contaminants.

In summary, oxic conditions in the hypolimnion are likely to:

- Substantially reduce the water column methylation in the hypolimnion and the subsequent transport of the methylmercury to the epilimnion via mixing.

- Shift the oxic/anoxic boundary from the water column to the sediment.

- Increase bioavailability of contaminants to the food chain via the interaction of benthic invertebrates with the sediments.

It is, however, unclear what the net effect of a shift to oxic conditions in the lake would be. While the concentrations of certain contaminants might decrease under oxic conditions in the hypolimnion, they would not be eliminated from the sediments. As a result, risks to benthic invertebrates, fish, and wildlife could increase substantially due to increased uptake and bioavailability of contaminants.

11.2.4 Eutrophication

The eutrophic nature of Onondaga Lake is due to elevated concentrations of ammonia, nitrite, phosphorus, and sulfide, resulting in depleted DO and reduced water transparency. However, measures are currently underway to improve wastewater treatment. Upgrades to the Metropolitan Syracuse Sewage Treatment Plant (Metro) are being guided by an Amended Consent Judgment (ACJ) from 1998 and decreases in effluent concentrations have been made in the last several years (e.g., Matthews et al., 2001). Under the ACJ, Onondaga County is to reduce stressors in Metro effluent over two intervals by December 2012.

Although future improvements are expected to lessen eutrophication in the lake, some COCs may become more bioavailable and could have a larger impact on the overall health of the ecosystem, as metals and synthetic organic chemicals in the sediments have much greater environmental persistence than stressors. Thus, the relative importance of COCs as compared to stressors of concern (SOCs) could increase in the future. In addition, unlike bioaccumulative metals and organic chemicals, nutrients may only affect the organisms that are directly exposed to them, and not wildlife at higher trophic levels.

11.3 Selection of Chemicals and Stressors of Concern

COC selection followed the procedures laid out in USEPA ERAGS (1997a) and subsequent guidance (USEPA, 2001). Screening-level exposure estimates and risk calculations were used to select COCs (see Chapter 6 and Appendix D). Stressors such as ionic waste (i.e., calcium, chloride, sodium), nutrients (i.e., nitrite, phosphorous, sulfide), salinity, ammonia, depleted DO, and reduced water transparency were used generally for qualitative evaluations and therefore were not screened in the same manner as COCs.

The COCs for water in Onondaga Lake and its tributaries were based on a screening evaluation applied to the results of chemical analyses conducted on water samples collected in the lake and its tributaries during the RI sampling in 1992 and 1999 and a review of the recent (1997 to 2001) OCDWEP water quality data. Given the amount of data available for review, as well as the conservativeness of the water quality standards, criteria, and guidance used for the screening evaluation (Chapter 4, Tables 4-3 and 4-4), it is probable that the list of COCs for water is complete.

COCs selected for the surface sediments of Onondaga Lake were based on a screening evaluation applied to the 1992 and 2000 lake sediment sampling data (see Appendix D, Tables D-14 to D-17 and D-52 to D-55). Given the relatively large amounts of data from multiple sources, the lakewide coverage of the sediment data sets, and the conservativeness of the sediment screening values used for the screening evaluation (Chapter 4, Tables 4-5 and 4-6), it is highly likely that the list of COCs for lake sediment is complete.

Surface soil/sediment COCs for the wetlands around Onondaga Lake and the dredge spoils area were based on a screening evaluation using the 2000 RI sampling data. Full TAL/Target Compound List (TCL) analyses were performed on samples from all areas using both soil and sediment screening values (Chapter 4, Table 4-7).

Plant screening values were available for only a subset of contaminants (mainly metals; Chapter 4, Table 4-8), so that although it is likely that the list of COCs for wildlife receptors exposed via terrestrial pathways is complete, the list of plant COCs may be incomplete.

The COCs identified for Onondaga Lake fish were based on a screening evaluation using the 1992 to 2000 Honeywell and NYSDEC sampling data. In the absence of fish screening values, wildlife (consumer) screening values were used (Chapter 4, Table 4-9), which are generally more protective. Given the relatively large amount of data from multiple sources, the range of species covering different trophic levels, and the conservative screening values used, it is highly likely that the list of COCs for fish is complete.

Food-web exposures were calculated using measured and modeled contaminant concentrations in surface water, sediment, and prey to select COCs on a receptor-specific basis. Maximum concentrations were used and conservative receptor parameters, such as minimum weight and maximum ingestion rates, were selected to maximize exposure. Therefore, it is highly likely that the list of COCs selected for avian and mammalian receptors is complete. A small subset of the COCs selected may be attributable to background or reference levels of contaminants, as discussed in the following section.

11.4 Background and Reference Concentrations

In keeping with USEPA policy (USEPA, 2002), this BERA retained constituents that exceeded risk-based screening concentrations. This evaluation of background and references concentrations evaluates the uncertainty associated with whether COCs are likely to be site-related. The contribution of background concentrations to risks is discussed separately for each medium in this section, based on available data for water, sediment, soil, and fish.

Estimations of background COC concentrations were based on the most appropriate data available in the Onondaga Lake database. Reference sampling stations associated with Geddes Brook and Ninemile Creek (i.e., Stations GB-2, NM-2, TN-17 [excluding TN-17-1A], and TN-18 [excluding TN-18-1A] [Exponent, 2001e; under revision]) were selected for comparison of contaminant concentrations to water, sediment, soil, and fish measured at Onondaga Lake. These stations are upstream of the Honeywell LCP Bridge Street site. Samples from Otisco Lake were also used as reference samples for sediment comparisons.

11.4.1 Reference Water Concentrations

Onondaga Lake surface water samples were compared to surface water samples collected from Geddes Brook/Ninemile Creek reference stations (i.e., GB-2 and NM-2). Concentrations of COCs in lake and reference water are provided in Table 11-1.

Most of the COCs selected for Onondaga Lake water were detected at substantially higher concentrations in lake water than reference surface water using both 95 percent upper confidence limit (UCL) or maximum concentrations and mean concentrations (Table 11-1). For example:

- Methylmercury concentrations were up to 17 times greater than reference concentrations.

- Total mercury concentrations were up to nine times greater than reference levels.

- Lead concentrations were up to seven times higher in lake water than reference samples.

- Organic COCs, zinc, and dissolved mercury were not detected in reference samples, but were detected in Onondaga Lake water.

- Barium had mean and upper-bound ratios ranging from 0.9 to 1.2, and is considered to be a reference COC in water.

- Manganese mean and upper-bound ratios were 2.3 and 1.8, respectively.

Based on these comparisons, all surface water COCs selected for Onondaga Lake, with the exception of barium and possibly manganese, can be considered site-related.

11.4.2 Reference Sediment Concentrations

Onondaga Lake surface sediment samples were compared to surface sediments collected from Otisco Lake and the upper Geddes Brook/Ninemile Creek reference stations (i.e., GB-2, NM-2, TN-17 [excluding TN-17-1A], and TN-18 [excluding TN-18-1A]). Concentrations of COCs in lake and reference sediments are provided in Table 11-2. Onondaga Lake and Otisco Lake sediment concentrations are based on the combined 1992 (0 to 2 cm) and 2000 (0 to 15 cm) data set, while the Geddes Brook/Ninemile Creek concentrations are based on the 1998 and 2001 (0 to 15 cm) sampling.

Two depths of Onondaga Lake sediments were compared to reference levels: the 1 m depth contour and the 9 m depth contour. Only receptors feeding on adult forms of aquatic invertebrates (i.e., tree swallow [*Tachycineta bicolor*] and little brown bat [*Myotis lucifugus*]) were modeled using the 9 m depth contour. Ratios of Onondaga Lake sediment concentrations to reference concentrations are provided in Table 11-3. Contaminants were considered elevated if either the upper bound or mean concentration was more than twice that of any of the reference locations. The use of a factor of two screening level does not mean that there is no toxic effect associated with those concentrations. The purpose of this factor is to concentrate on those COCs that are the most significant contributors to the ecological risk and are most likely site-related.

Concentrations of arsenic, copper, manganese, and vanadium in the 1 m and 9 m contours of Onondaga Lake sediments were within a factor of two for both Geddes Brook/Ninemile Creek and Otisco Lake sediments (Table 11-3). Zinc was within a factor of two for all comparisons except the Onondaga Lake-Otisco Lake 9 m contour comparison, where it was 2.1.

Levels of antimony, selenium, and DDT and metabolites were within a factor of two of Otisco Lake stations and were not detected in Geddes Brook/Ninemile Creek.

Silver, benzene, dichlorobenzenes, ethylbenzene, toluene, trichlorobenzenes, total xylenes, hexachlorobenzene, phenol, and dieldrin were not detected at either of the reference locations. Ratios of the remaining COCs that were greater than those of background levels by at least an order-of-magnitude are as follows:

- Total PCBs were detected at more than an order-of-magnitude higher in Onondaga Lake than at reference stations.

- Mercury, methylmercury, and dioxin/furans (avian toxicity equivalent [TEQ]) were detected at more than two orders-of-magnitude higher than at reference stations.

- Total PAHs in Onondaga Lake sediments were detected at more than three orders-of-magnitude higher than at reference stations.

Based on these comparisons, sediment COCs considered to be site-related are: barium, cadmium, chromium, lead, mercury, methylmercury, nickel, and all organic COCs (with the exception of DDT and metabolites).

Sediment concentrations were used for some receptors to model benthic invertebrate concentrations in prey. Receptors with a major sediment-based component of their diet are the tree swallow, mallard, and little brown bat. Receptor COCs with HQs above 1.0 that may be attributable to background risks are:

- Tree swallow (Table 10-5) – selenium and zinc.
- Mallard (Table 10-6) – zinc.
- Little brown bat (Table 10-11) – arsenic, copper, and vanadium.

The remaining HQs are considered to be site-related. Receptors deriving the majority of their food from prey other than aquatic invertebrates are discussed in the following sections.

11.4.3 Background and Reference Soil Concentrations

Mean and 95 percent UCL surface soil concentrations in wetlands and dredge spoils were compared to upper Geddes Brook/Ninemile Creek reference stations (i.e., GB-2 and NM-2) (Table 11-4). The most suitable background numbers for the wetlands would be samples from similar type wetlands. In the absence of that data, the reference station and background soil values were used, but are not ideal. Ratios comparing Onondaga Lake wetland and dredge spoils area concentrations to reference station concentrations were calculated (Table 11-5). Contaminants were considered elevated if either the upper bound or mean concentration was more than twice that of reference locations. Naturally occurring inorganic elements detected in Onondaga Lake soil that were not detected at the reference stations were also compared to the average concentrations provided in "Background of 20 Elements in Soils with Special Regard to New York State" (McGovern, nd).

The results of the reference location and background comparisons were considered together to determine whether inorganic COCs were site-related. For the four wetland areas and the dredge spoils area, the results were as follows:

- Wetland SYW-6 (northwest end of the lake) had elevated concentrations of mercury, cadmium, chromium, lead, nickel, selenium, zinc, cyanide, total PCBs, and dioxins/furans, compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. The elevated concentrations of PAHs detected in the NYSDEC/TAMS 2002 Wetland SYW-6 sample were in the 15 to 30 cm interval, which is below the interval used in this BERA (0 to 15 cm).

- Wetland SYW-10 (near the mouth of Ninemile Creek) had elevated concentrations of arsenic, cadmium, lead, mercury, hexachlorobenzene, total PCBs, and dioxins/furans as compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. Wetland SYW-10 will be evaluated further as part of the Geddes Brook/Ninemile Creek site RI/FS.

- Wetland SYW-12 (southeast end of the lake) had elevated concentrations of cadmium, chromium, lead, mercury, chlorobenzene, dichlorobenzenes, hexachlorobenzene, selenium, and total PCBs as compared to upper Geddes Brook/Ninemile Creek reference stations and background levels.

- Wetland SYW-19 (southwest end of the lake) had concentrations of mercury and PCBs over an order-of-magnitude above levels at the reference stations. Concentrations of barium, cadmium, lead, selenium, silver, and all organic contaminants (i.e., benzene, chlorobenzene, dichlorobenzenes, trichlorobenzenes, hexachlorobenzene, phenol, aldrin, dieldrin, total PAHs, total PCBs, and dioxins/furans) were also elevated compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. Wetland SYW-19 will be evaluated further as part of the Wastebed B/Harbor Brook site RI/FS.

- In the surface soils of the dredge spoils area ,concentrations of all COCs, with the exception of mercury, silver, dichlorobenzenes, and hexachlorobenzene were comparable to upper Geddes Brook/Ninemile Creek reference stations and/or average background concentrations found in the literature. The dredge spoils area will be evaluated further as a separate site with its own investigation.

Based on this comparison, soil COCs considered to be site-related at one or more wetland area include: arsenic, barium, cadmium, chromium, lead, mercury, nickel, selenium, silver, zinc, cyanide, and all organic COCs (i.e., benzene, chlorobenzene, dichlorobenzenes, trichlorobenzenes, hexachlorobenzene, phenol, aldrin, dieldrin, total PAHs, and total PCBs). All mercury was considered to be site-related, based on known releases to the lake.

Soil concentrations were used for some receptors for direct comparisons (plants) or to model concentrations in terrestrial invertebrate prey (short-tailed shrew [(*Blarina brevida*)]) or small mammals (red-tailed hawk [*Buteo jamaicensis*] and mink [*Mustela vison*]). Receptor COCs with HQs above 1.0 that may be attributable to background risks are:

- Plants (Table 10-1) – selenium (Wetland SYW-10 and dredge spoils), vanadium (all locations), and zinc (all locations except Wetland SYW-6).

- Red-tailed hawk (Table 10-10) – no COCs attributable to background risks.

- Short-tailed shrew (Table 10-12) – selenium (Wetland SYW-10 and dredge spoils)and vanadium (all locations).

- Mink (Table 10-13) – no COCs attributable to background risks.

11.4.4 Reference Fish Concentrations

Reference sampling stations used to estimate reference fish concentrations were based on selected sampling stations (i.e., GB-2 and NM-2) associated with the Geddes Brook and Ninemile Creek RI (Table 11-6). Six white sucker (*Catostomus commersoni*), one creek chub (*Semotilus atromaculatus*), and one tessellated darter (*Etheostoma nigrum*) were collected at these stations.

The white sucker feeds on a variety of organisms occurring in the mud, including aquatic insect larvae, small mollusks, and crustaceans in stream and lake bottoms. The white sucker serves as prey for other fish, including walleye (*Stizostedion vitreum*), largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieui*), and other game fish. Because of similar feeding strategies, contaminant concentrations in the white sucker were compared to carp (*Cyprinus carpio*) and catfish contaminant concentrations measured in Onondaga Lake.

The white sucker reference station samples had HQs above 1.0 for arsenic, chromium, selenium, vanadium, and zinc when the maximum concentration was compared to the no observable adverse effect level (NOAEL) (Table 11-6). Selenium and vanadium exceeded the NOAEL at both mean and maximum concentrations. All exceedances were within one order-of-magnitude.

Concentrations of contaminants in carp and catfish from Onondaga Lake fish were up to two orders-of-magnitude greater than concentrations of contaminants measured in reference station fish (Table 11-7). Ratios of site concentrations to reference concentrations were highest for bioaccumulative organic contaminants, such as endrin and DDT and metabolites. Concentrations of mercury, total PCBs, chromium, selenium, and vanadium were also substantially higher in Onondaga Lake fish.

The creek chub and tessellated darter were compared to the bluegill (*Lepomis macrochirus*) to evaluate reference concentrations, as both the creek chub and tessellated darter feed on benthic invertebrates (Table 11-7). Creek chub and tessellated darter reference samples were analyzed for only a limited number of contaminants (mercury, PCBs, dioxins/furans), none of which had HQs greater than one (Table 11-6). Concentrations of mercury in lake fish were over an order-of-magnitude greater than in reference fish and concentrations of dioxins/furans were also higher in Onondaga Lake fish (Table 11-7).

Based on these ratios of COC HQs, all COCs in fish and receptors feeding primarily on fish (belted kingfisher [*Ceryle alcyon*], great blue heron [*Ardea herodias*], osprey [*Pandion haliaetus*], mink [*Mustela vison*], and otter [*Lutra canadensis*]) are considered to be site-related.

11.5 Sediment Effect Concentrations and Macroinvertebrate Uncertainties

11.5.1 Representativeness of Toxicity Tests

The two test species used to assess sediment toxicity during the 1992 and 2000 RI sampling were the amphipod *Hyalella azteca* and the chironomid *Chironomus tentans*. Both species are standard test organisms, but their true sensitivity to sediment contamination levels is uncertain. In addition, the two species represent two different major taxonomic groups (crustaceans and insects, respectively) and occupy different positions in the sediment. *H. azteca* tends to live on the sediment surface, whereas *C. tentans* lives in a case it constructs within the sediment. The joint use of the these two test species ensured that a range of taxa and exposure scenarios was evaluated.

In addition to using two different test species, a variety of exposure conditions and toxicity endpoints were used to assess sediment toxicity in Onondaga Lake. In 1992, the test species were exposed to the top 2 cm of field-collected sediments for ten days under static conditions, with toxicity endpoints of survival and biomass. In 2000, the test species were exposed to the top 15 cm of field-collected sediment for 42 days with renewal of overlying water, with toxicity endpoints of survival and biomass for both species, number of young for the amphipods, and emergence for the chironomids. Comparisons of the 1992 and 2000 results showed that the spatial patterns of toxicity identified by both sets of data were similar, indicating that sediment toxicity in the lake was adequately characterized.

The principal uncertainties surrounding toxicity tests are that while they are a good measure of the potential for adverse environmental effects by providing indications of whether conditions are toxic enough to kill or otherwise impact test species, they do not exactly mimic natural exposure. As a consequence, there is some degree of uncertainty in relating the toxicity test results directly to the potential for actual responses in Onondaga Lake. Laboratory toxicity tests were used to measure the acute or chronic toxicity of site sediment samples to test organisms. They were unable to definitively determine if the sediment was also toxic to lake biota in-situ or to determine which COCs or SOCs were the specific cause of the toxicity. The rationale for conducting toxicity tests on Onondaga Lake sediments was to provide a quantifiable measure of the potential for the occurrence of effects.

Sediment toxicity tests automatically take into account the relative toxicity of a mixture of chemicals, including any synergistic or antagonistic effect between chemicals. However, the ability to determine a direct causative link to one or more contaminant in the sediment may be highly uncertain due to the presence of many co-occurring contaminants. If toxic effects are found and there is no correlation between the effects and the contamination levels, the measured toxicity could be the result of an unanalyzed substance or other substances such as ammonia or sulfides. Because the sediment toxicity in much of Onondaga Lake is a result of the exposure of organisms to a very complex mixture of metals and organic contaminants, it is generally difficult to correlate the toxicity at any given site to any particular contaminant.

11.5.2 Selection of Sediment Effect Concentrations

The sediment screening values used for the general screening evaluation were based on the lowest available values. The site-specific probable effect concentrations (PECs) used for the evaluations of potential risk posed by individual sediment samples were based on an approach developed by Ingersoll et al. (1996 and 2000). Because this approach was applied to the extensive amount of data collected during the RI on sediment chemical concentrations and associated sediment toxicity in Onondaga Lake, it has considerably more site-specific relevance than do generic sediment screening values. As shown in Table 11-8, Onondaga Lake PECs are generally more conservative than those previously published (Ingersoll et al., 2000), with PEC values anywhere from one-half the value to ten times lower than values previously published. However, the Onondaga Lake PEC for mercury (2.2 mg/kg) is twice the Ingersoll value (1.1 mg/kg).

Despite the site-specific applicability of the PECs, uncertainties exist with respect to their ability to identify which COCs may be responsible for causing any observed toxicity. This uncertainty is present with most kinds of sediment quality values found in the literature and is the result of potential confounding factors that are often encountered in environmental samples (e.g., co-occurring chemicals, site-specific factors that modify bioavailability, heterogeneous sediment matrices). In addition, no PECs based on field information can conclusively identify causal relationships. Instead, associations between chemicals and biological effects are used to infer potential causative relationships. This lack of conclusive causality is a source of uncertainty in using PECs for sediment assessment.

11.5.3 Uncertainties Associated with the Use of Benthic Metrics

The benthic data were evaluated using five metrics (i.e., species richness, dominance index, abundance of indicator species, community composition, and species diversity). The cumulative review of the five metrics (referred to as a "multi-metric approach") was used to coalesce the metrics into a single overall assessment of each station. There are uncertainties associated with the use of the benthic metrics in this manner, as follows:

- The level of impact, or "impairment" (i.e., non-impaired, slightly impaired, moderately impaired, or severely impaired), that is derived from the metrics cannot be attributed directly to specific COCs.

- A level of uncertainty exists due to the fact that all five metrics rarely show the same impairment level at a given station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment.

- Benthic invertebrate communities typically occur in patches in the natural environment. To account for this when sampling, replicate samples were collected to improve sampling precision. In this study, five replicates were collected at each station. While this is a good effort (three replicates are considered the absolute minimum for macroinvertebrate characterization), there is an inherent uncertainty about whether enough replicates were taken to obtain meaningful estimates.

11.5.4 Uncertainties Associated with the Simultaneously Extracted Metals/Acid-Volatile Sulfide Ratios

Although it has been shown through numerous laboratory experiments that consideration of simultaneously extracted metals/acid-volatile sulfide (SEM/AVS) ratios can improve predictions of sediment toxicity due to divalent metals, there are uncertainties associated with the approach. The approach has been largely tested for acute toxicity and, therefore, has uncertain applicability to chronic toxicity. In addition, the predictive ability of this approach for sediments in a stratified lake such as Onondaga Lake is uncertain because it has been demonstrated that sediment AVS concentrations can vary temporally and spatially.

In general, AVS concentrations tend to increase during periods of stratification and decrease at times of the year when the water column becomes oxygenated. The AVS may, therefore, limit the bioavailability of divalent metals for only part of the year. Use of AVS data collected from the time of year when AVS levels are expected to be highest could bias the data interpretation for the remaining part of the year. This is due to the seasonal oxidation conditions at the sediment-water interface, so that metals sequestered during one season may be released during another. In addition, the SEM/AVS approach should not be used to characterize mercury toxicity, even though mercury forms sulfide complexes, because the organic form of mercury is the most bioavailable and toxic form and does not complex with sediment sulfides. However, it should also be pointed out that AVS may limit the availability of inorganic mercury to methylate in lake sediments.

11.6 Conceptual Model Uncertainties

The conceptual model links COC sources, likely exposure pathways, and potential ecological receptors. It is intended to provide broad linkages from various receptor groups found in and around Onondaga Lake to contamination in Onondaga Lake water, sediments, soils, and prey. The conceptual model has been refined since its initial presentation in the Onondaga Lake Work Plan (PTI, 1991). Based on changes made to the model as more was learned about the Onondaga Lake ecosystem, there is considered to be a low level of uncertainty associated with the conceptual model. However, since it is a generalized model, it is not intended to represent specific individuals currently living around Onondaga Lake. The actual linkages between the biotic levels depend on the seasonal availability of various prey and food items.

The results of the risk characterization show that the majority of risk is due to exposure to contaminated prey, which is consistent with other studies. Specific uncertainties in the exposure and food-web modeling are discussed in the following section.

11.7 Natural Variation and Parameter Error

Natural variation represents known variation in parameters based on observed heterogeneity in the characteristics of a particular receptor species. Variability can often be reduced with additional data collection, whereas uncertainty can be reduced directly through the confirmation of applied assumptions or inferences through direct measurement. Parameter error includes both uncertainty in estimating specific parameters related to exposure or the specific exposure point concentrations (EPCs) being applied in the exposure models (e.g., sediment, water, and fish concentrations, etc.) as well as variability (e.g., ingestion rate, body weight, temporal and spatial habitat use, etc.). Some parameters can be both uncertain and variable.

11.7.1 Receptor Exposure Parameters

11.7.1.1 Body Mass

Body mass plays a quantitative role in the water, dietary, and incidental sediment ingestion pathways as part of the average daily dosage term for each pathway on a per-kilogram body weight basis. Body masses for adult birds were generally based on mean or median body masses provided in references such as Dunning (1993) and USEPA (1993b), in contrast to the screening-level risk assessment where minimum body weights were used. Representative mammalian body masses were taken from North American populations. Measurements for regional populations, such as New York populations of tree swallows and little brown bats, were used when available.

On a cumulative dosage basis, a higher body mass estimate would reduce the estimated cumulative daily dosage fraction of COCs on a per-kilogram body weight basis. Likewise, a lower body mass estimate would result in a higher average daily dosage estimate. Since it is not known if typical body masses for Onondaga Lake populations are indicative of either extreme in the range of body masses, no systematic bias is associated with these estimates. Therefore, body masses employed in the exposure pathway modeling for avian and mammalian receptors are considered reliable and representative of Onondaga Lake populations.

11.7.1.2 Ingestion Rates

Food Ingestion Rates

Estimates of food ingestion rates (FIRs) for all receptors, with the exception of the short-tailed shrew, were derived using the bioenergetic allometric scaling function of Nagy (1987). This function relates field metabolic rates to body mass across receptors within a given class (birds or mammals). The bioenergetic algorithm of Nagy (1987) did not include data for very small, very active eutherian mammals, such as the shrew. Since the field metabolic rate is strongly correlated with body size, it was considered inappropriate to use Nagy's equation to calculate a metabolic rate for shrews, and literature ingestion rates were used instead to estimate intake. The little brown bat is also a small, active mammal; however, as it spends part of the year in hibernation, the Nagy equation was used to estimate a year-round intake rate.

Use of allometric scaling incorporates some degree of uncertainty in the absence of field verification. To reduce this uncertainty, diet-normalized metabolic rates and the metabolizable energy contents of specific foods consumed were used. Ingestion rates were calculated as the quotient of the species-specific normalized metabolic rate and the average metabolizable energy content of the diet. Estimation of the average gross energy content in wildlife foods is limited to a number of select broad phylogenic groups and is rarely available for species-level evaluations of prey included in the diet. Reliance upon the gross energy estimates for representative taxa groups introduces some uncertainty in derivation of the ingestion rates, as it is assumed that the gross energy content and assimilative efficiency of select groups of invertebrates and fish taxa are equivalent to other freshwater benthic invertebrate and fish taxa. This assumption in the energy content of the diet can influence the ingestion rate estimate, if under- or overestimated. An overestimate of the average metabolizable energy in the diet will decrease the ingestion rate (i.e., the actual metabolic average is lower than estimated), while an underestimate of the metabolic average results in an overestimate of the ingestion rate. There was no systematic bias inherent in the FIRs used in this BERA.

Water Ingestion Rates

Water ingestion rates (WIRs) for avian and mammalian receptors were estimated based upon allometric relationships developed for mammals and birds by Calder and Braun (1983). For this pathway, it was assumed that avian and mammalian receptors use Onondaga Lake as their exclusive drinking water source. The dosage estimate for water ingestion did not account for metabolic- or dietary-derived sources of water for the individual receptors. Consequently, the allometric methods assumed that hydration demands in the receptors are solely accounted for by direct ingestion of surface water. This assumption may result in a slight overestimate of surface water-derived COC exposure through the drinking water pathway by exclusion of metabolic and dietary sources.

Incidental Sediment Ingestion Rates

Of the receptors evaluated, only the mallard and short-tailed shrew have published estimates for ingestion of soil/sediment. The value of 3.3 percent for the mallard (Beyer et al., 1994) and 13 percent for the short-tailed shrew (Talmage and Walton, 1993) are quantified estimates and are considered reliable for application to Onondaga Lake populations.

Estimates of incidental sediment ingestion for other receptors were made based upon feeding behavior used for prey capture and consumption and nesting/resting habitats of each species. Both the tree swallow and little brown bat feed primarily on flying insects that are captured and consumed in flight. The tree swallow nests in trees, while the little brown bat roosts in sheltered locations, such as caves and abandoned buildings. These feeding and roosting preferences result in incomplete pathways for incidental sediment ingestion. Therefore, a zero percent incidental sediment ingestion rate (SIR) was used for both receptors.

The great blue heron, belted kingfisher, and osprey were characterized as primarily piscivorous in diet. All three receptor species visually follow their prey and seize the specific prey item using their bill (great blue heron and belted kingfisher) or talons (osprey). An SIR of 1 percent was used for the great blue heron as it may ingest some incidental sediment during prey capture, prey consumption, and grooming. This rate was also applied to the belted kingfisher, which has little contact with sediments during feeding, but may ingest some sediments during grooming because it nests in riverbanks. An SIR of zero percent was applied to the osprey based on its feeding and nesting habits.

Stomach content and scat analyses on mink from New York State have shown trace quantities (i.e., less than or equal to 1 percent of the diet) of sand present (Hamilton, 1940). Based upon this study and the potential for the mink to also ingest sediments during grooming, a 1 percent incidental ingestion composition in the diet of the mink was applied.

No quantitative dietary information regarding the occurrence of soils/sediments in the diet of the river otter was available, but based on the potential to ingest sediments during feeding and grooming, a 1 percent ingestion rate was also applied to the river otter.

The 1 percent SIR does not consider sediment contained in the digestive system of prey. A study evaluating the stomach contents of bluegills reported that an average of 9.6 percent of the bluegill's diet consisted of detritus and sediment (Kolehmainen, 1974). The majority of the fish data used in this BERA were based on whole-fish samples, which include sediment contained in the stomach. Whole fish to fillet conversion factors were applied to fish that were analyzed as fillet samples, when appropriate. Therefore, the incidental SIR of piscivorous receptors is considered appropriate, as incidental sediment contained in fish prey is included in fish exposure concentrations.

11.7.2 Temporal and Spatial Parameters

11.7.2.1 Uncertainty in Temporal Parameters

A year-round exposure time was used for avian and mammalian receptors in this BERA. Although the avian receptors considered in this BERA are migratory in nature, there have been year-round sightings of them at Onondaga Lake (see Chapter 8, Section 8.2). Even if receptors migrate, they are likely to breed and raise their young at Onondaga Lake during warmer periods of the year. As reproductive effects were generally selected as toxicity endpoints, full-time residency may slightly overestimate exposure; however this is considered to be appropriate.

The mammalian receptors selected in this assessment are year-round residents of Onondaga Lake. However, the little brown bat hibernates during the winter. During hibernation it relies on food reserves obtained during the summer and fall. As food reserves are obtained from Onondaga Lake, it is considered to have year-round exposure.

11.7.2.2 Uncertainty in Spatial Parameters

The conceptual model assumes that receptors modeled belong to closed populations that forage exclusively in and around Onondaga Lake. While this may be accurate for receptors with small home ranges (e.g., belted kingfisher, short-tailed shrew), exposure may be overestimated for receptors with larger home ranges (e.g., osprey, river otter). Prey availability plays a major role in the home range and location of receptors. During years with low prey availability, some receptors may obtain a portion of their food off-site, while during years with high prey abundance all food may come from Onondaga Lake. Therefore, the uncertainty in the spatial use of the site may introduce a conservative bias in some years.

11.8 Model Error

A food-web model was used to approximate relationships between site-specific environmental conditions (i.e., exposure sources) and receptors. Relationships between trophic levels and food-web components are well understood, but available models are generally simplistic. Potential sources of error in the model are discussed below.

11.8.1 Prey Contaminant Exposure Concentrations

The evaluation of the 95 percent UCL exposure assumes that exposure via multiple routes is at the 95 percent UCL for all media. In the calculation of the total exposure, the summation of the exposure route concentrations at the 95 percent UCL of the mean could result in estimates that may be higher than the actual 95 percent UCL exposure concentrations. However, as the majority of risk in food-chain models is derived from dietary intake (e.g., fish or invertebrates as prey), the use of multiple 95 percent UCL exposure concentrations is not considered overly conservative. In addition, mean exposure concentrations are calculated to provide a range of HQs for each receptor.

11.8.1.1 Uncertainty in Chemical of Concern Exposure Concentrations in Fish

Estimates of COC concentrations in fish (as both prey and receptor) were derived from direct measurements. There were a limited amount of data for TAL metals and some organic contaminants, particularly in the 3 to 18 cm range, but data were generally sufficient to perform the statistical tests (i.e., determination of data distribution type and calculation of the 95 percent UCL on the mean). The use of maximum concentrations as the upper-bound estimate for contaminants with few size-class or species-specific samples available may introduce a conservative bias into these estimates, specifically in the 95 percent UCL estimate for receptors that eat small fish (see Appendix I, Tables I-5 to I-14 for upper-bound exposure concentrations used). However, sufficient mercury data were available in both the 3 to 18 cm and 18 to 60 cm fish size ranges to calculate a 95 percent UCL.

Fillet to Whole Fish Conversion Factors

Conversion factors were used to adjust mercury, DDT and metabolites, PCBs, and dioxin/furan concentrations in fillets to whole-body concentrations. The factors used for mercury and PCBs have a high degree of confidence based on the large number of site-specific data and comparison to literature values. The DDT and dioxin/furan values are based on smaller data sets and have a moderate degree of uncertainty associated with them, based upon the generalized assumption that fillet to whole-body relationships were independent of species and age. The remaining conversion factors calculated (see Table 8-4) were considered to have a high degree of uncertainty associated with them, and were, therefore, not used in this assessment. There is no systematic bias in the fillet to whole-fish conversion factors.

11.8.1.2 Uncertainty in Chemical of Concern Exposure Concentrations in Plant and Non-Fish Prey Sources

Concentrations of COCs were not measured in terrestrial plants, terrestrial invertebrates, birds (including eggs), mammals, or benthic macroinvertebrates, with the exception of mercury and PCBs in benthic macroinvertebrates. Therefore, COC concentrations in these prey types were estimated based on assumed media-transfer relationships. The uncertainty in media-transfer ratios and functional relationships between COC concentrations in sediment (or soil) and in tissue is greater than that of using measured prey concentrations.

Biota-sediment accumulation factors for aquatic invertebrates and uptake factors (UFs) or equations for earthworms and small mammals were taken from reports published by Oak Ridge National Laboratory (ORNL) (Sample et al. 1998a,b; US Department of Energy [USDOE], 1998). General, rather than conservative, estimates were applied to reduce the level of conservatism associated with these estimates. The COC-specific factors in these publications are based on linear or transformed functions or point estimates, of transfer coefficients derived from a survey of available literature data. There were situations where no COC-specific transfer relationship could be found and the estimate of the concentrations in tissue relative to a medium had to be based on a surrogate contaminant for which a transfer coefficient was available.

For functional relationships that represent contaminant transfer from the environment to tissue, such as those applied in Sample et al. (1998a,b) and USDOE (1998), the regression relationships are derived from a database pooled from various studies. The underlying assumption is that observations from independent studies represent random and unbiased estimates of the same relationship, and that all variances between such observations are experimental in nature and not the result of differences in experimental design and/or approach. Transfer of contaminants to body tissue is a multi-functional and dynamic process dependent on factors such as duration of exposure, availability from the medium, depuration rates, receptor species, health status, and habitat type. The uncertainty resulting from this approach cannot be quantified based on the data available from either the literature or Onondaga Lake sampling, but is not considered to be biased in either direction. Omitting pathways that lack site-specific data could substantially underestimate risk.

11.9 Toxicological Uncertainties

Uncertainties in toxicological studies may result from the use of laboratory or field studies that may differ from the actual toxicity present at Onondaga Lake due to:

- Site-specific conditions.
- Interspecies differences in sensitivity to contaminants.
- Extrapolating between lowest observed adverse effect level (LOAEL) and the NOAEL, and vice-versa.
- Extrapolating from subchronic to chronic exposures.
- Actual bioavailability of contaminants.

Risk prediction is dependent upon the assumption that daily exposure to COC doses greater than the toxicity reference value (TRV) will result in an adverse effect. Toxicological studies showing reproductive effects were preferred when available, as reproductive effects are considered to be a sensitive endpoint. However, the actual impact of reproductive effects on receptor populations possesses some uncertainty with regard to magnitude of ecological impact relative to predicted risk. Because the level of impact is based upon a physiological rather than an ecological TRV, the uncertainty tends to be conservative. The range of toxicity thresholds reported in the literature is large, even among those studies deemed suitable for extrapolation to the receptor species of interest. The range may be due to test species, life stage, exposure dosage and duration, the form of a contaminant, or other factors.

11.9.1 Laboratory Versus Field Studies

Both laboratory and field studies have advantages and disadvantages with respect to use in the development of TRVs. Laboratory experiments offer the advantage of being able to control exposure conditions, while field studies may more closely represent actual exposure conditions. For example, the concentrations of contaminants in environmental media, especially tissue, may be strongly influenced by differential rates of transport, uptake, metabolism, and elimination. Contaminants that are resistant to metabolism are more persistent and tend to be present at higher concentrations in environmental media than in a commercial mixture (e.g., Aroclor 1254). Therefore, forms of contaminants in environmental media (e.g., fish tissue or bird eggs) may be more toxic than laboratory mixtures, and TRVs based on laboratory dietary doses may underestimate the toxicity of the dietary dose received by a receptor in the field.

Laboratory studies are often designed to test the effect of a single contaminant on a test species in the absence of other co-occurring contaminants, and, thus, observed effects are clearly related to exposure to the test compound. In field studies, organisms are typically exposed to other co-occurring contaminants. The presence of co-occurring contaminants may be a disadvantage to the use of field studies for development of TRVs, since observed effects may not be solely attributable to exposure to a specific contaminant. Laboratory studies were used to derive most TRVs in this BERA, except when a more appropriate field study was available.

11.9.2 Interspecies Sensitivity

Species often vary in their sensitivity to contaminants. An interspecies uncertainty factor estimates differences in sensitivity, but the test species could be either more or less sensitive than the receptor of concern. Certain taxonomic groups of animals, such as salmonid fish, gallinaceous birds, and mink have been shown to be highly sensitive to the reproductive effects of certain contaminants, such as PCBs (e.g., Beyer et al., 1996). To minimize this source of uncertainty in the BERA, studies on sensitive receptors were only selected when alternative studies were not available, excluding receptors classified as sensitive species. Analysis of the available literature provided no reason to assume that the receptors evaluated in this investigation would be more or less sensitive to the COCs than those tested in the respective toxicity studies selected, unless noted in the text. Therefore, any variance in the sensitivity of the receptor relative to the test species used to develop the TRV would most likely be evenly distributed around the estimated TRV, and no interspecies uncertainty factors were applied.

11.9.3 Application of Conversion Factors

Additional areas of uncertainty are encountered when the best available study for the development of a final TRV uses a subchronic, rather than a chronic, exposure. A conversion factor of 0.1 is used to estimate a chronic TRV from a subchronic TRV. A conversion factor differs from an uncertainty factor in that the direction of the uncertainty is known. For example, the chronic TRV is expected to be lower than the subchronic TRV. Use of a subchronic-to-chronic conversion factor of 0.1 is supported by the results of a study that compared subchronic to chronic NOAELs and LOAELs (Dourson and Stara, 1983). For more than half of the chemicals studied, the ratio of subchronic to chronic endpoints was 2.0 or less, and for 96 percent of the chemicals the ratios were below 10. Therefore, application of a conversion factor of 10 was considered protective and may result in a slight conservative bias for the following TRVs where a subchronic-to-chronic conversion factor was applied:

- Chromium – Mammalian NOAEL and LOAEL.
- Dichlorobenzenes – Avian NOAEL and LOAEL.

Uncertainty also exists when conversion factors are used to estimate NOAELs from LOAELs. Data on the ratio of LOAEL to NOAEL indicates that all chemicals examined have a LOAEL to NOAEL ratio of 10 or less and 96 percent have a ratio of five or less (Dourson and Stara, 1983). Therefore, a factor of 10 was used to convert between NOAELs and LOAELs. The direction of uncertainty associated with the use of a LOAEL to NOAEL conversion factor is known, since NOAELs are always expected to be lower than LOAELs. LOAEL to NOAEL conversion factors were used for the following TRVs:

- Arsenic – Mammalian NOAEL.
- Total mercury – Mammalian LOAEL.
- Methylmercury – Avian and mammalian NOAELs.
- Thallium – Mammalian NOAEL.
- Vanadium – Avian LOAEL and mammalian NOAEL.
- Total PCBs – Avian and mink/otter NOAELs.
- Total PAHs – Avian and mammalian NOAELs.
- Zinc – Fish NOAEL.

11.9.4 Uncertainty in Relative Bioavailability

The bioaccumulation and response models (for both plants and animals) assumed that the form of the chemical present in the environment was absorbed with the same efficiency as the chemical form used in the laboratory toxicity study. Chemical solubility is an important factor in absorption efficiency, and for many chemicals, laboratory toxicity studies are performed using the most soluble form. This is particularly true of the metal COCs, which are themselves natural but often biologically unavailable constituents of abiotic media such as soils and sediments.

Concentrations of COCs in the soils surrounding Onondaga Lake were analyzed using USEPA Method 3050b extractions, which rely on digestion using nitric acid and hydrogen peroxide under high temperature to solubilize the metal constituents. Metals may be more available using this method than in their natural state, where they are usually covalently bound within the soil matrix. The assumption that the concentrations measured from matrices that have undergone strong acid digestion represents the fraction available for uptake by plants or absorption by animals is a conservative assumption. Although this method provides a conservative estimate of plant risk, the resulting data are the only available estimate of contaminants in soil and are therefore used to determine whether concentrations of COCs measured in soil may pose a risk to plants.

11.9.5 Uncertainty Due to Lack of Appropriate Toxicity Data

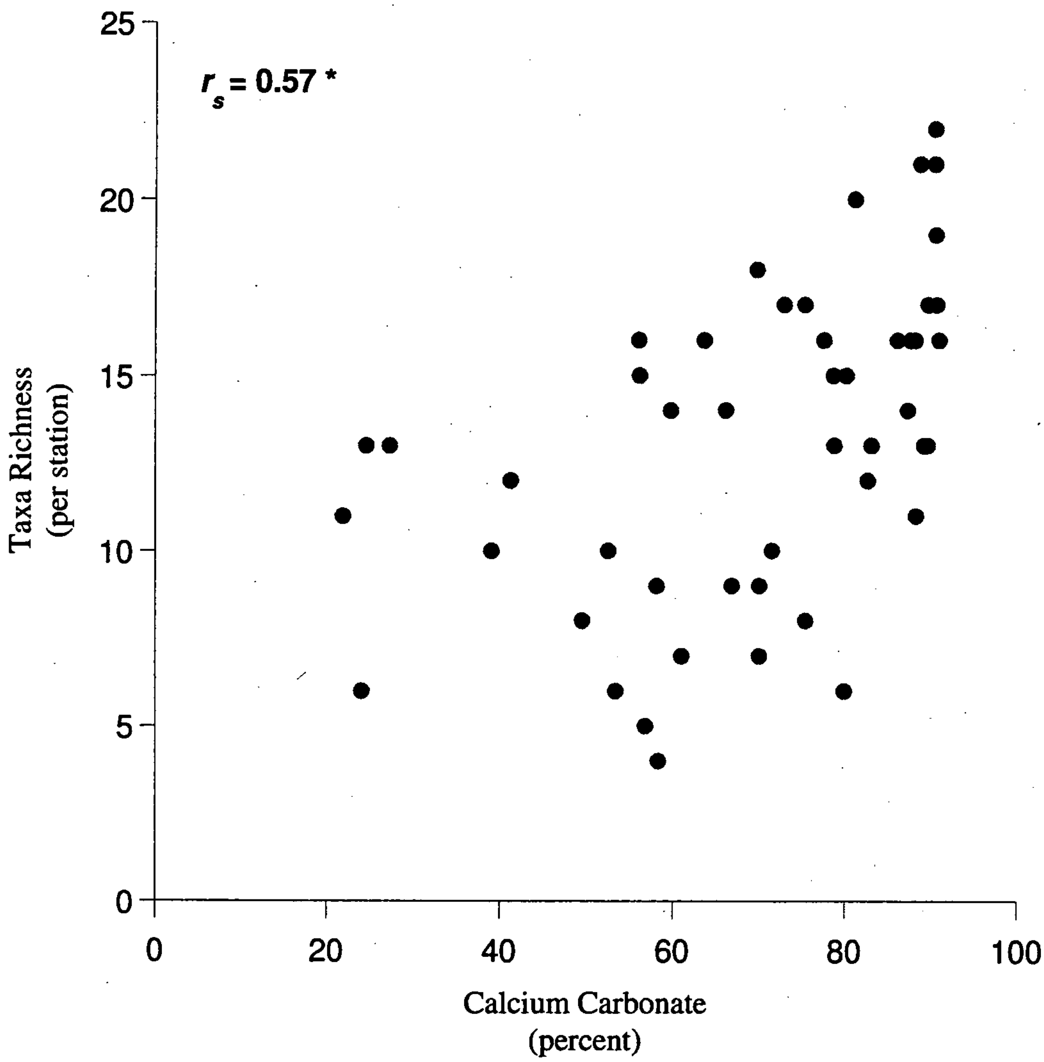
Appropriate toxicity studies were not available for avian receptors for thallium, trichlorobenzenes, and xylenes. It was therefore not possible to calculate risks for these COCs to avian receptors, potentially underestimating risks.

Other toxicity studies measured endpoints, such as mortality, which are generally less sensitive than reproductive endpoints and may underestimate risks to receptors. Survival of eggs to juvenile life stages were grouped together with reproductive effects. Studies used to derive TRVs for this BERA based on adult survival endpoints are:

- Antimony – fish and birds.
- Chromium – fish.

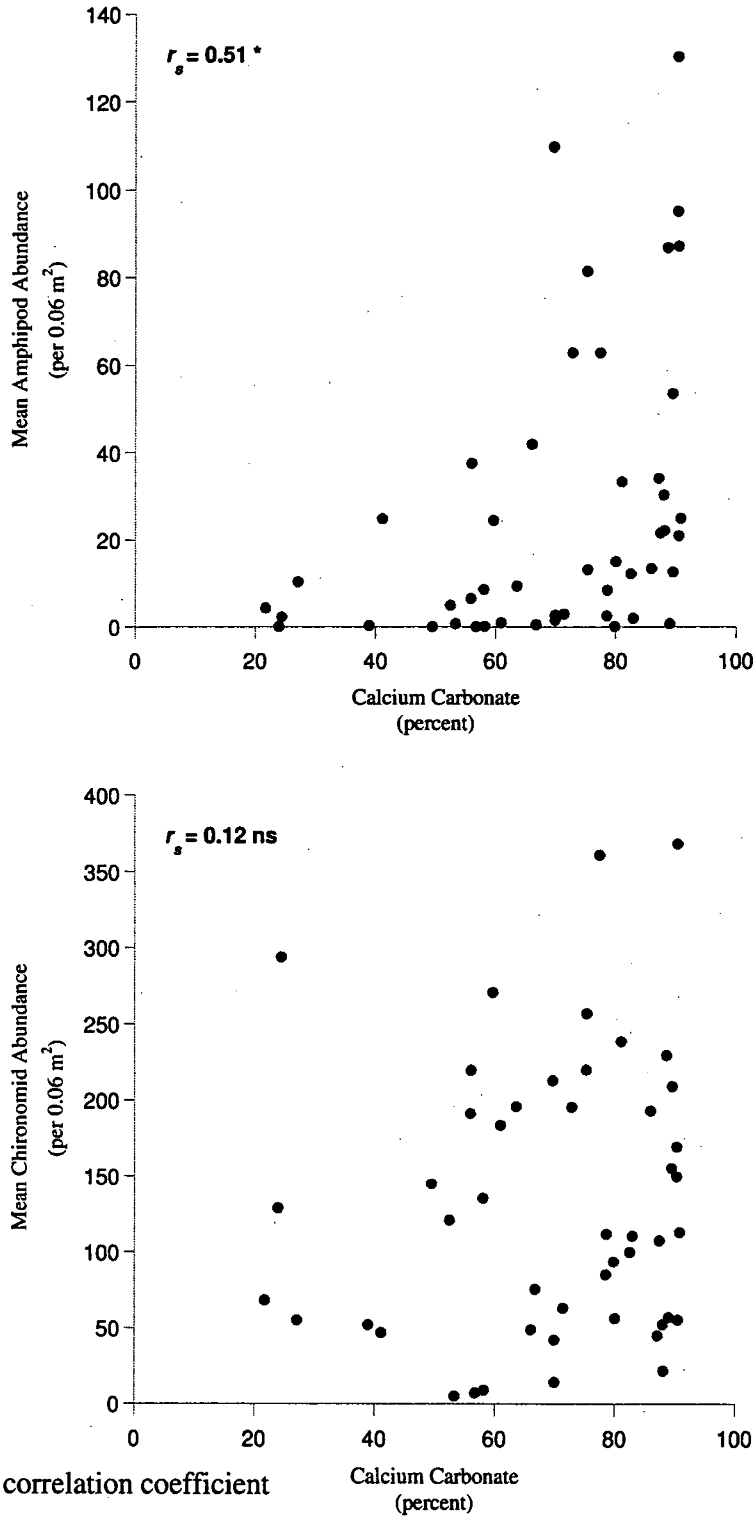
11.10 Summary

Uncertainty is an inherent component of risk assessments. Elements of uncertainty in this BERA have been identified and efforts have been made to minimize them. For components in which a moderate degree of uncertainty is unavoidable (e.g., sampling data), efforts have been made, to the extent possible, to minimize any systematic bias associated with the data.



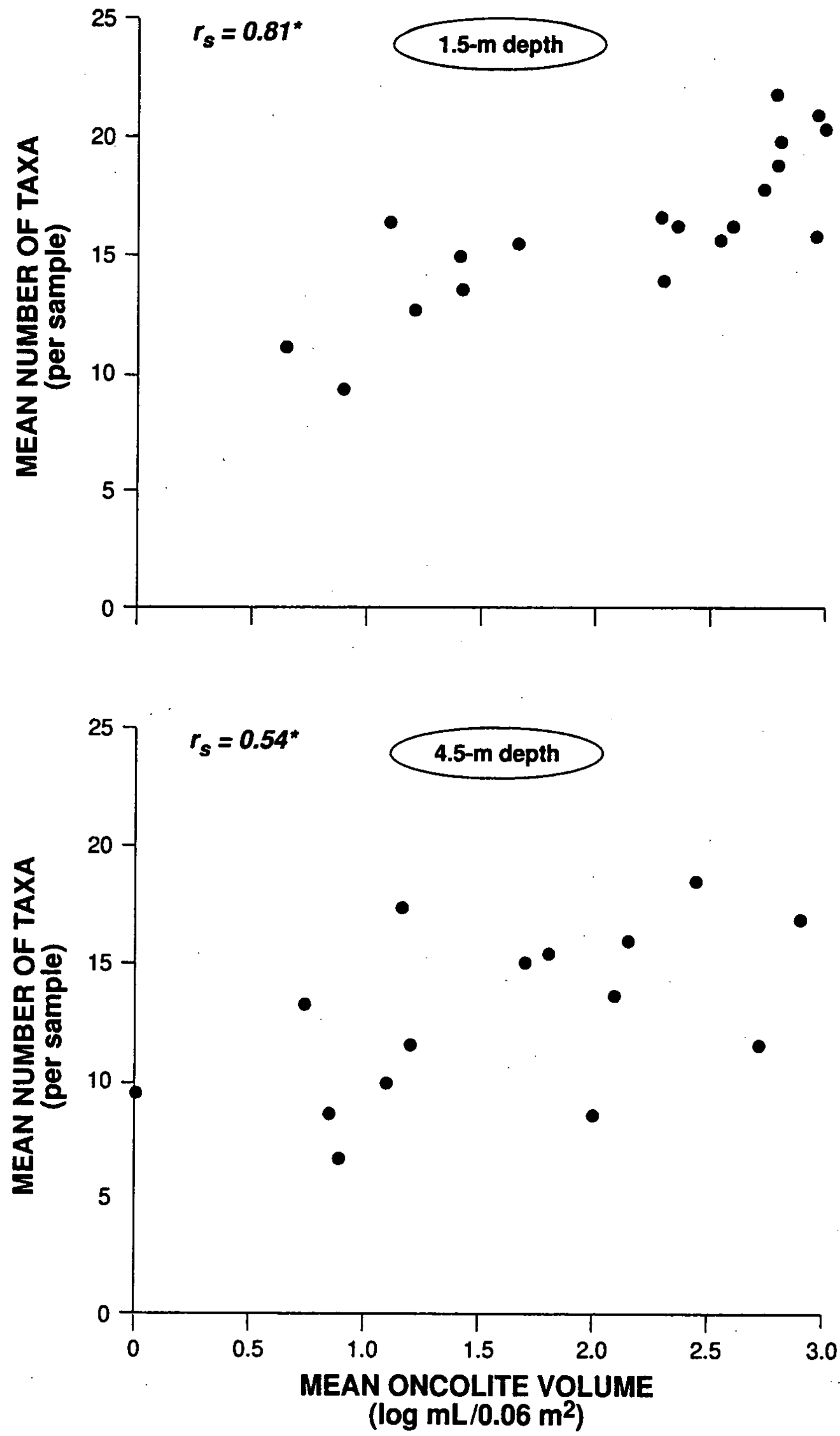
Note: r_s Spearman rank correlation coefficient
* $P \leqq 0.05$

Figure 11-1. Comparison of Taxa Richness with Calcium Carbonate Content of Sediment in Onondaga Lake in 1992



Note: r_s Spearman rank correlation coefficient
* P ≦ 0.05
ns P > 0.05

Figure 11-2. Comparison of Amphipod and Chironomid Abundances with Calcium Carbonate Content of Sediment in Onondaga Lake in 1992

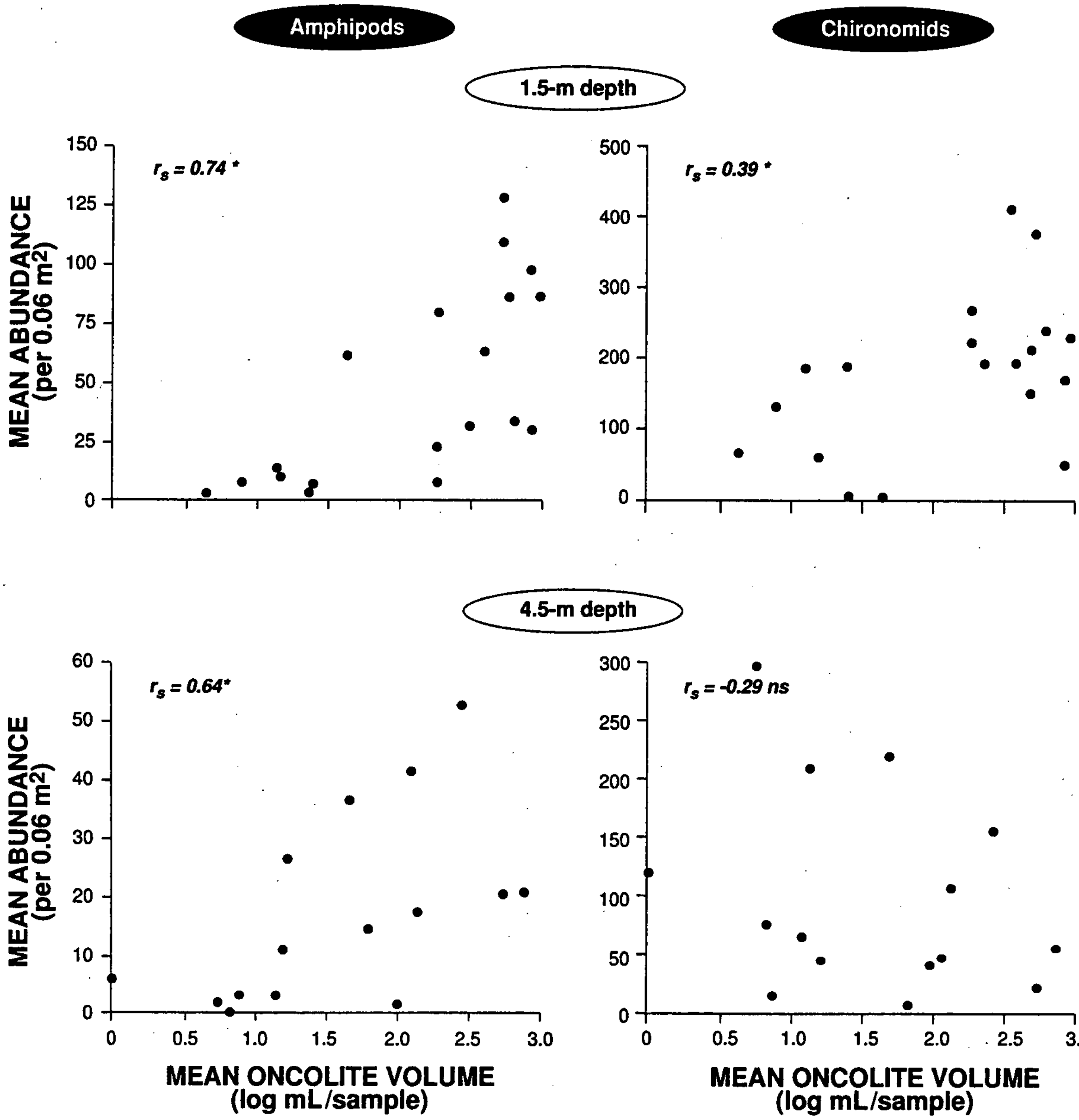


Note: r_s Spearman rank correlation coefficient

* $P \leqq 0.05$

Source: Exponent, 2001b

Figure 11-3. Comparison of taxa richness with oncolite volume for water depths of 1.5 m and 4.5 m in Onondaga Lake in 1992



Note: r_s Spearman rank correlation coefficient

* $P \leqq 0.05$; ns = $P > 0.05$

Source: Exponent, 2001b

Figure 11-4. Comparison of amphipod and chironomid abundances with oncolite volume for water depths of 1.5 m and 4.5 m in Onondaga Lake in 1992

Table 11-1. Ratios of Onondaga Lake Surface Water COCs to Reference Location Concentrations

COC	Units	Reference UCL (max)	Reference Mean	Lake 95% UCL	Lake Mean	UCL Ratio Lake:Reference	Mean Ratio Lake:Reference
Barium*	µg/L	73	56	68	67	0.9	1.2
Copper	µg/L	4.8	2.7	11	9.1	2.3	3.3
Lead	µg/L	1.5	1.0	6.9	6.4	4.6	6.6
Manganese	µg/L	50	36	112	65	2.3	1.8
Mercury-dissolved	ng/L-dis	ND	ND	2.7	2.3	ND BKGD	ND BKGD
Mercury-total	ng/L	6.4	3.4	36	29	5.7	8.7
Methylmercury-dissolved	ng/L-dis	3.8E-02	2.9E-02	0.6	0.4	17	14
Zinc	µg/L	ND	ND	65	45	ND BKGD	ND BKGD
Chlorobenzene	µg/L	ND	ND	0.6	0.6	ND BKGD	ND BKGD
Dichlorobenzene (Sum)	µg/L-dis	ND	ND	1.3	1.7	ND BKGD	ND BKGD
Trichlorobenzenes (Sum)	µg/L-dis	ND	ND	0.6	0.6	ND BKGD	ND BKGD
Bis(2-ethylhexyl)phthalate*	µg/L-dis	ND	ND	10	6.0	ND BKGD	ND BKGD

Notes:

ND - not detected

ND BKGD- not detected in background samples and therefore no ratio could be calculated.

Reference locations used as background samples consist of Stations NM2 and GB2 from 1998 Geddes Brook/Ninemile Creek RI sampling.

* Barium and BEHP samples are taken from 6 m depth since no 1 m depth samples were available.

All other COC concentrations are based on 1 m depth samples.

UCL – upper confidence limit

Table 11-2. Surface Sediment Concentrations in Onondaga Lake and Reference Locations

COC	Onon Lake 1-m contour 95%UCL	Onon Lake 1-m contour Mean	Onon Lake 9-m contour 95%UCL	Onon Lake 9-m contour Mean	Upper GB/NMC 95%UCL	Upper GB/NMC Mean	Otisco Lake 95%UCL	Otisco Lake Mean
Metals (mg/kg)								
Antimony	1.5	1.1	1.3	1.0	ND	ND	0.8	0.8
Arsenic	8.4	5.2	7.6	5.4	4.7	4.0	7.7	4.1
Barium	405	320	392	330	82	66	189	102
Cadmium	2.2	1.4	3.3	2.2	ND	ND	0.7	0.2
Chromium	139	130	158	169	32	23	24	11
Copper	66	48	66	53	34	28	158	73
Lead	116	74	98	72	50	40	32	16
Manganese	313	287	342	318	355	296	1180	686
Mercury	11	5.4	12	7.1	0.8	0.3	0.2	0.1
Methylmercury (μg/kg)	29	7.9	30	94	2.1	0.8	1.4	0.9
Nickel	52	49	53	60	19	17	23	15
Selenium	1.1	0.9	1.2	1.0	ND	ND	3.2	0.8
Silver	0.8	0.6	1.4	0.9	ND	ND	ND	ND
Vanadium	23	14	30	21	17	17	24	11
Zinc	102	87	123	107	103	89	84	52
Volatile Organic Compounds (μg/kg)								
Benzene	11,776	1,645	2,289	1,068	ND	ND	ND	ND
Chlorobenzene	476,553	18,775	67,689	13,473	ND	ND	ND	ND
Dichlorobenzenes (Sum)	37,046	5,879	11,919	5,562	ND	ND	ND	ND
Ethylbenzene	7,831	1,466	1,587	979	ND	ND	ND	ND
Toluene	8,300	893	2,174	645	ND	ND	NA	NA
Trichlorobenzenes (Sum)	1,147	1,721	578	1,277	ND	ND	ND	ND
Total Xylenes	330,000	11,814	32,989	6,825	ND	ND	ND	ND
Semivolatile Organic Compounds (μg/kg)								
Dibenzofuran	3,354	3,806	1,599	2,767	73	194	ND	ND
Hexachlorobenzene	1,768	505	484	374	ND	ND	ND	ND
Phenol	1,326	2,362	911	1,780	ND	ND	ND	ND
Total PAHs	1,293,496	387,587	224,725	227,949	30,039	16,965	1,026	367

Table 11-2. (cont.)

COC	Onon Lake 1-m contour 95%UCL	Onon Lake 1-m contour Mean	Onon Lake 9-m contour 95%UCL	Onon Lake 9-m contour Mean	Upper GB/NMC 95%UCL	Upper GB/NMC Mean	Otisco Lake 95%UCL	Otisco Lake Mean
Pesticides/Polychlorinated Biphenyls (µg/kg)								
Chlordane (Sum)	2.8	3.2	3.1	3.3	2.8	2.2	1.2	1.8
DDT and metabolites (Sum)	12	8.3	11	7.9	ND	ND	13	4.7
Dieldrin	3.2	2.6	4.1	3.3	ND	ND	ND	ND
Heptachlor /Hept. epoxide	3.3	3.3	3.4	3.2	2.3	2.0	1.4	1.8
PCBs (Sum)	629	490	704	646	ND	ND	124	51
Dioxins/Furans (ng/kg)								
TEQ (1/2 DL) Avian	524	119	524	117	3.3	2.6	NA	NA
TEQ (1/2 DL) Mammalian	165	43	165	44	2.7	1.7	NA	NA

Notes:

ND = not detected

NA= not analyzed

DDT – dichlorodiphenyltrichloroethane

PAHs –polycyclic aromatic hydrocarbons

PCBs –polychlorinated biphenyls

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 11-3. Ratios of Onondaga Lake Sediment (0-15 cm) COC Concentrations to Reference Locations

COC	Lake 1 m: GB/NMC 95%UCL	Lake 1m: GB/NMC Mean	Lake 1m: Otisco 95%UCL	Lake 1m: Otisco Mean	Lake 9 m: GB/NMC 95%UCL	Lake 9m: GB/NMC Mean	Lake 9m: Otisco 95%UCL	Lake 9m: Otisco Mean
Metals								
Antimony	BKGD ND	BKGD ND	1.8	1.3	BKGD ND	BKGD ND	1.6	1.3
Arsenic	1.8	1.3	1.1	1.3	1.6	1.3	1.0	1.3
Barium	4.9	4.8	2.1	3.2	4.8	5.0	2.1	3.2
Cadmium	BKGD ND	BKGD ND	3.0	6.2	BKGD ND	BKGD ND	4.5	9.7
Chromium	4.3	5.6	5.8	12	4.9	7.3	6.6	15
Copper	2.0	1.7	0.4	0.7	2.0	1.9	0.4	0.7
Lead	2.3	1.9	3.6	4.5	2.0	1.8	3.0	4.4
Manganese	0.9	1.0	0.3	0.4	1.0	1.1	0.3	0.5
Mercury	13	18	48	68	14	24	54	90
Methylmercury	14	10	21	9.2	15	126	22	110
Nickel	2.7	2.9	2.2	3.4	2.8	3.5	2.3	4.1
Selenium	BKGD ND	BKGD ND	0.4	1.1	BKGD ND	BKGD ND	0.4	1.2
Silver	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Vanadium	1.3	0.8	1.0	1.3	1.3	0.8	1.3	2.0
Zinc	1.0	1.0	1.2	1.7	1.2	1.2	1.5	2.1
Volatile Organic Compounds								
Benzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Chlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Dichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Ethylbenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Toluene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Trichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Total Xylenes	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Semivolatile Organic Compounds								
Dibenzofuran	46	20	BKGD ND	BKGD ND	22	14	BKGD ND	BKGD ND
Hexachlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Phenol	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Total PAHs	43	23	1261	1056	7.5	13	219	621

Table 11-3. (cont.)

COC	Lake 1 m: GB/NMC 95%UCL	Lake 1m: GB/NMC Mean	Lake 1m: Otisco 95%UCL	Lake 1m: Otisco Mean	Lake 9 m: GB/NMC 95%UCL	Lake 9m: GB/NMC Mean	Lake 9m: Otisco 95%UCL	Lake 9m: Otisco Mean
Pesticides/Polychlorinated Biphenyls								
Chlordane (Sum)	1.0	1.5	2.4	1.8	1.1	1.5	2.7	1.9
DDT and metabolites (Sum)	BKGD ND	BKGD ND	0.9	1.8	BKGD ND	BKGD ND	0.9	1.7
Dieldrin	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Heptachlor and Heptachlor epoxide	1.4	1.7	2.4	1.8	1.5	1.6	2.4	1.8
PCBs (Sum)	BKGD ND	BKGD ND	5.1	9.6	BKGD ND	BKGD ND	5.7	12.7
Dioxins/Furans								
TEQ (1/2 DL) Avian	157	46	BKGD NA	BKGD NA	157	46	BKGD NA	BKGD NA
TEQ (1/2 DL) Mammalian	61	25	BKGD NA	BKGD NA	61	26	BKGD NA	BKGD NA

Notes:

BKGD ND- Not detected in reference station samples.

BKGD NA- Not analyzed in reference station samples.

Reference stations serving as background sites are NM2, GB2, TN-17, and TN-18 from the Geddes Brook/Ninemile Creek 1998/2001 RI sampling and Otisco Lake 1992 and 2000 sampling.

DDT – dichlorodiphenyltrichloroethane

PAHs –polycyclic aromatic hydrocarbons

PCBs –polychlorinated biphenyls

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 11-4. Surface Soil Concentrations in Onondaga Lake, Reference Locations, and Background Literature

COC	Combined Soils 95%UCL	Combined Soils Mean	Dredge Spoils 95%UCL	Dredge Spoils Mean	SYW-6 95%UCL	SYW-6 Mean	SWY-10 95%UCL	SWY-10 Mean
Metals (mg/kg)								
Antimony	0.5	0.4	0.4	0.3	2.2	0.6	0.5	0.3
Arsenic	0.5	0.4	0.4	0.3	2.2	0.6	1	0.3
Barium	156	128	78	72	176	125	157	105
Cadmium	14	2.0	ND	ND	14	3	0.9	0.5
Chromium	51	39	29	17	154	49	47	27
Copper	57	42	24	17	120	46	49	35
Iron	12,973	11,443	17,100	13,808	24,000	10,170	21,600	13,223
Lead	106	60	14	11	175	72	115	59
Manganese	301	278	354	299	406	267	488	344
Mercury	18	3.0	4.0	0.6	4.5	1.3	3.4	2.1
Methylmercury (µg/kg)	0.2	4.0E-02	NA	NA	4.5E-02	1.3E-02	3.4E-02	2.1E-02
Nickel	28	23	17	14	64	29	34	20
Selenium	0.8	0.4	ND	ND	1.4	0.5	ND	ND
Silver	1.2	0.9	1.4	1.0	2.5	0.8	1.8	0.7
Vanadium	16	14	29	19	22	13	31	16
Thallium	0.8	0.6	ND	ND	1.4	0.6	2.5	1.5
Zinc	159	118	50	39	510	181	119	97
Cyanide	0.8	0.6	ND	ND	1.4	0.6	2.5	1.5
Volatile Organic Compounds (µg/kg)								
Benzene	13	8.7	NA	NA	NA	NA	ND	ND
Chlorobenzene	600	69	NA	NA	NA	NA	ND	ND
Dichlorobenzenes (Sum)	4,518	1,400	51	24	ND	ND	ND	ND
Trichlorobenzenes (Sum)	1,229	512	ND	ND	ND	ND	ND	ND
Semivolatile Organic Compounds (µg/kg)								
Hexachlorobenzene	4,255	395	410	69	ND	ND	35	26
PAHs (Sum)	184,400	13,289	1,541	425	22,450	6,245	17,202	5,227
Phenol	519	254	ND	ND	n/a	328	ND	ND
Pesticides/Polychlorinated Biphenyls (µg/kg)								
Chlordane (sum)	17	4.7	NA	NA	ND	ND	ND	ND
DDT and metabolites	50	12	NA	NA	2.3	1.5	3.5	1.6
Hexachlorocyclohexanes	4.3	2.1	NA	NA	ND	ND	ND	ND
Aldrin	31	6.8	NA	NA	ND	ND	ND	ND
Dieldrin	22	5.3	NA	NA	ND	ND	ND	ND
PCBs (Sum)	17	5	56	33	ND	ND	ND	ND
Dioxins/Furans (ng/kg)								
TEQ (1/2 DL) Avian	2,168	275	2.9	1.8	34	15	25	17
TEQ (1/2 DL) Mammalian	1,086	128	1.4	0.9	20	8.8	6.9	5.8

Table 11-4. (cont.)

COC	SYW-12 95%UCL	SYW-12 Mean	SYW-19 95%UCL	SYW-19 Mean	Upper GB/NMC 95%UCL	Upper GB/NMC Mean	Background Mean
Metals (mg/kg)							
Antimony	0.6	0.3	1.1	0.6	ND	ND	ND
Arsenic	0.6	0.3	1.1	0.6	6.4	4.7	5.0
Barium	152	98	390	302	93	70	290
Cadmium	8.8	5.3	2.3	1.3	ND	ND	0.2
Chromium	115	66	55	43	36	22	33
Copper	88	49	167	85	83	42	13
Iron	11,800	8,763	11,750	10,478	23,300	17,267	14000
Lead	116	77	259	118	56	44	17
Manganese	284	239	303	233	581	433	345
Mercury	1.5	0.7	25	15	0.52	0.24	0.8
Methylmercury (μg/kg)	1.5E-02	1.1E-02	0.3	0.1	3.31	1.75	ND
Nickel	32	19	44	32	26	19	20
Selenium	2.7	1.2	1.3	0.5	ND	ND	0.8
Silver	0.9	0.4	1.7	1.4	ND	ND	0.01-8
Vanadium	16	9	13	12	20	17	43
Thallium	ND	ND	ND	ND	ND	ND	ND
Zinc	241	160	138	114	179	136	64
Cyanide	ND	ND	ND	ND	ND	ND	ND
Volatile Organic Compounds (μg/kg)							
Benzene	NA	NA	60	18	ND	ND	ND
Chlorobenzene	2.0	2.8	600	199	ND	ND	ND
Dichlorobenzenes (Sum)	54	75	14,700	9,258	ND	ND	ND
Trichlorobenzenes (Sum)	ND	ND	6,550	2,838	ND	ND	ND
Semivolatile Organic Compounds (μg/kg)							
Hexachlorobenzene	31	10	5,355	1,972	ND	ND	ND
PAHs (Sum)	20,480	7,830	184,400	68,387	37,324	27,878	ND
Phenol	ND	ND	2,825	965	ND	ND	ND
Pesticides/Polychlorinated Biphenyls (μg/kg)							
Chlordane (sum)	8.5	4.7	30	13	3.1	1.8	ND
DDT and metabolites	9.7	6.8	56	39	14	6.4	ND
Hexachlorocyclohexanes	1.7	0.8	10	7	ND	ND	ND
Aldrin	ND	ND	45	24	ND	ND	ND
Dieldrin	5.0	2.8	24	17	ND	ND	ND
PCBs (Sum)	9	5	30	13	80	20	ND
Dioxins/Furans (ng/kg)							
TEQ (1/2 DL) Avian	NA	NA	2,168	1,066	3.4	2.6	ND
TEQ (1/2 DL) Mammalian	NA	NA	1,086	498	3.2	1.9	ND

Notes: ND-denotes Not Detected; NA- denotes Not Analyzed.
GB/NMC reference sites include Stations NM2 and GB2.
Background means based on McGovern (no date) and Kabata-Pendias and Pendias (1984).
DDT – dichlorodiphenyltrichloroethane
PAHs – polycyclic aromatic hydrocarbons
PCBs – polychlorinated biphenyls
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 11-5. Ratios of Onondaga Lake Soil Concentrations to Geddes Brook/Ninemile Creek Reference Stations

COC	Combined Soils 95%UCL	Combined Soils Mean	Dredge Spoils 95%UCL	Dredge Spoils Mean	SYW-6 95%UCL	SYW-6 Mean
Metals						
Antimony	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Arsenic	1.0	1.0	1.3	1.1	0.9	0.6
Barium	1.7	1.8	0.8	1.0	1.9	1.8
Cadmium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Chromium	1.4	1.8	0.8	0.8	4.3	2.2
Copper	0.7	1.0	0.3	0.4	1.5	1.1
Iron	0.6	0.7	0.7	0.8	1.0	0.6
Lead	1.9	1.4	0.2	0.3	3.1	1.6
Manganese	0.5	0.6	0.6	0.7	0.7	0.6
Mercury	35.4	12.6	7.7	2.7	8.5	5.5
Methylmercury	NA	NA	NA	NA	NA	NA
Nickel	1.1	1.2	0.6	0.7	2.5	1.5
Selenium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Silver	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Vanadium	0.8	0.9	1.4	1.1	1.1	0.8
Thallium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Zinc	0.9	0.9	0.3	0.3	2.8	1.3
Cyanide	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Volatile Organic Compounds						
Benzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Chlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Dichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Trichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Semivolatile Organic Compounds						
Hexachlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Total PAHs	4.9	0.5	4.1E-02	1.5E-02	0.6	0.2
Phenol	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Pesticides/Polychlorinated Biphenyls						
Chlordane (Sum)	5.4	2.6	NA	NA	ND	ND
DDT and metabolites	3.6	1.9	NA	NA	0.2	0.2
Hexachlorocyclohexanes	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Aldrin	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Dieldrin	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Total PCBs	5.7	11	0.7	1.7	0.8	2.4
Dioxins/Furans (ng/kg)						
TEQ (1/2 DL) Avian	630	105	0.8	0.7	9.9	5.7
TEQ (1/2 DL) Mammalian	338	66	0.5	0.5	6.2	4.5

Table 11-5. (cont.)

COC		SYW-19 95%UCL	SYW-19 Mean	SYW-12 95%UCL	SYW-12 Mean	SYW-10 95%UCL	SYW-10 Mean
Metals							
	Antimony	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Arsenic	1.3	1.5	0.6	0.5	2.9	1.5
	Barium	4.2	4.3	1.6	1.4	1.7	1.5
	Cadmium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Chromium	1.5	1.9	3.2	3.0	1.3	1.2
	Copper	2.0	2.0	1.1	1.2	0.6	0.8
	Iron	0.5	0.6	0.5	0.5	0.9	0.8
	Lead	4.6	2.7	2.1	1.7	2.1	1.3
	Manganese	0.5	0.5	0.5	0.6	0.8	0.8
	Mercury	48	62	2.8	2.8	6.4	8.9
	Methylmercury	NA	NA	NA	NA	NA	NA
	Nickel	1.7	1.7	1.2	1.0	1.3	1.1
	Selenium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Silver	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Vanadium	0.6	0.7	0.8	0.5	1.5	0.9
	Thallium	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Zinc	0.8	0.8	1.3	1.2	0.7	0.7
	Cyanide	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Volatile Organic Compounds							
	Benzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Chlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Dichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Trichlorobenzenes (Sum)	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Semivolatile Organic Compounds							
	Hexachlorobenzene	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Total PAHs	4.9	2.5	0.5	0.3	0.5	0.2
	Phenol	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
Pesticides/Polychlorinated Biphenyls							
	Chlordane (Sum)	9.6	7.1	2.7	2.6	ND	ND
	DDT and metabolites	4.0	6.1	0.7	1.1	0.2	0.3
	Hexachlorocyclohexanes	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Aldrin	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Dieldrin	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND	BKGD ND
	Total PCBs	13	45	4.2	12	2.0	4.2
Dioxins/Furans (ng/kg)							
	TEQ (1/2 DL) Avian	630	406	NA	NA	7.3	6.6
	TEQ (1/2 DL) Mammalian	338	256	NA	NA	2.1	3.0

Notes: BKGD ND- Not detected in reference station samples.
Reference stations serving as background sites are NM2 and GB2 from the Geddes Brook/Ninemile Creek 1998 RI sampling.
DDT – dichlorodiphenyltrichloroethane
PAHs –polycyclic aromatic hydrocarbons
PCBs –polychlorinated biphenyls
TEQ – toxicity equivalence quotient
UCL – upper confidence limit

Table 11-6. Hazard Quotients of Reference Location Fish

COC		White Sucker 95%UCL HQ NOAEL	White Sucker 95%UCL HQ LOAEL	White Sucker Mean HQ NOAEL	White Sucker Mean HQ LOAEL HQ	Creek Chub Max NOAEL HQ	Creek Chub Max LOAEL HQ	Tessellated Darter Max NOAEL HQ	Tessellated Darter Max LOAEL HQ
Antimony	mg/kg-ww	NA	NA	NA	NA	NA	NA	NA	NA
Arsenic	mg/kg-ww	1.1	0.4	0.6	0.2	NA	NA	NA	NA
Chromium	mg/kg-ww	2.2	0.7	0.9	0.3	NA	NA	NA	NA
Mercury	mg/kg-ww	1.5E-01	7.5E-02	9.6E-02	4.8E-02	4.1E-02	2.0E-02	3.7E-02	1.9E-02
Methylmercury	mg/kg-ww	NA	NA	NA	NA	NA	NA	NA	NA
Selenium	mg/kg-dw	2.2	0.218	1.1	0.11	NA	NA	NA	NA
Vanadium	mg/kg-ww	7.6	0.761	2.9	0.29	NA	NA	NA	NA
Zinc	mg/kg-ww	1.2	0.116	0.954	0.10	NA	NA	NA	NA
Endrin	mg/kg-ww	6.0E-02	6.0E-03	6.0E-02	6.0E-03	ND	ND	NA	NA
DDT and metabolites	mg/kg-ww	7.2E-03	1.5E-03	6.2E-03	1.3E-03	NA	NA	NA	NA
PCBs	mg/kg-ww	0.9	0.2	0.6	0.1	ND	ND	NA	NA
TEQ (1/2 DL) Fish	µg/kg-lipid	0.7	0.4	0.3	0.1	NA	NA	0.2	7.4E-02

Notes:

ND - not detected

NA- not analyzed

All samples are from Station NM2 in Ninemile Creek.

Sample sizes of reference fish:

White sucker n= 6

Creek chub n=1

Tessellated darter n=1.

DDT – dichlorodiphenyltrichloroethane

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCBs –polychlorinated biphenyls

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 11-7. Ratios of Onondaga Lake Fish COC Concentrations to Reference Locations

COC	95%UCL Lake Carp:Bkgd WS	95%UCL Lake Cfish:Bkgd WS	Mean Lake Carp:Bkgd WS	Mean Lake Cfish:Bkgd WS	95%UCL Lake Bgill:Bkgd CChub	95%UCL Lake Bgill:Bkgd TDarter
Antimony	NA	NA	NA	NA	NA	NA
Arsenic	3.6	NA	2.8	NA	NA	NA
Chromium	9.5	8.7	7.6	20	NA	NA
Mercury	5.7	17	7.4	26	27	29
Methylmercury	NA	NA	NA	NA	NA	NA
Selenium	9.2	59	7.4	111	NA	NA
Vanadium	3.2	35	4.5	91	NA	NA
Zinc	17	29	10	35	NA	NA
Endrin	80	612	37	612	NA	NA
DDT and metabolites	50	386	46	449	NA	NA
Total PCBs	3.0	12	2.6	17	NA	NA
TEQ (1/2 DL) Fish	3.5	0.8	3.9	1.4	NA	2.9

Notes:

NA - Not available

White sucker (WS) n=6

Creek chub n=1

Tesselated darter n=1.

DDT – dichlorodiphenyltrichloroethane

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCBs –polychlorinated biphenyls

TEQ – toxicity equivalence quotient

UCL – upper confidence limit

Table 11-8. Comparison of Various Site-Specific Sediment Effect Concentrations for Onondaga Lake, 1992 Data[a,b]

	AET	ER-L	ER-M	TEL	PEL	**Onondaga Lake Consensus PECs**	**Ingersoll et al., 2000 PECs**
Metals (mg/kg)							
Arsenic	4.3	0.90	4.4	1.29	3.55	**2.4**	**33**
Cadmium	8.6	0.94	2.1	1.42	3.11	**2.4**	**5.0**
Chromium	195	17.6	47.9	29.3	67.3	**50**	**111**
Copper	83.7	12.3	40.7	19.1	48.3	**33**	**149**
Lead	116	9.68	56.9	13.3	57.6	**35**	**128**
Total mercury	13	0.51	2.8	0.99	2.84	**2.2**	**1.1**
Nickel	50	5.22	20.9	8.37	25.8	**16**	**49**
Zinc	218	37.9	94.6	56.7	120	**88**	**459**
Organic Compounds (μg/kg)							
Total PCBs	710	136	400	151	382	**295**	**676**
PAH Compounds (μg/kg)							
Naphthalene	2,100	340	1,400	471	1,380	**917**	**561**
Fluorene	3,500	55.2	305	66.9	327	**264**	**536**
Phenanthrene	16,000	92.2	480	135	491	**542**	**1,170**
Anthracene	4,400	33	210	49.6	249	**207**	**845**
Fluoranthene	26,000	140	1,400	483	2,482	**1,436**	**2,230**
Pyrene	NC	114	650	238	795	**344**	**1,520**
Benz[a]anthracene	NC	60.7	415	118	451	**191**	**1,050**
Chrysene	NC	100	440	172	541	**253**	**1,290**
Benzo[a]pyrene	NC	62.8	210	98.2	355	**146**	**1,450**
Pesticides (μg/kg)							
DDT and Metobolites	16.3	47	47	23.7	26.6	**30**	**572**
Chlordane	NC	NC	NC	5.08	5.08	**5.1**	**18**
Heptachlor and Heptachlor Epoxide	NC	NC	NC	NC	NC		**16**

Notes: [a] All concentrations are provided in dry weight.

[b] Maps of exceedances of ER-L, ER-M, TEL, PEL and PEC values are presented in Appendix F.

AET - apparent effects threshold; ER-L - effects-range low; ER-M - effects-range median
TEL - threshold effect level; PEL - probable effect level; PEC - Probable Effect Concentration
BTX - benzene, toluene, xylenes; PCB - polychlorinated biphenyl; PAH - polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
NC - value was not calculated because of an insufficient number of detected observations.

12. CONCLUSIONS

This chapter summarizes the results of the baseline ecological risk assessment presented in the preceding chapters. Each assessment endpoint and its associated measurement endpoints are presented in a strength-of-evidence approach along with a summary of the results. A strength-of-evidence approach is used to integrate different types of data, or lines of evidence used in this BERA to support a conclusion. The results of the risk characterization are evaluated in the context of the uncertainty analysis (Chapter 11) to assess the potential for adverse effects to receptors as a result of exposure to contaminants and stressors present in Onondaga Lake.

12.1 Assessment Endpoint: Sustainability of an Aquatic Macrophyte Community That Can Serve as a Shelter and Food Source for Local Invertebrates, Fish, and Wildlife

Does the Aquatic Macrophyte Community Structure Reflect the Influence of Chemicals of Concern/Stressors of Concern (COCs/SOCs)?

Studies of the Onondaga Lake macrophyte community, as compared to reference lakes, indicate that the current impoverished community does reflect the influence of COCs and SOCs, particularly ionic waste. Lower species diversity is seen than in similar lakes, and macrophyte coverage of the lake is low.

Do the COCs/SOCs Present in Onondaga Lake Affect Macrophyte Growth and Survival?

Laboratory (greenhouse studies) and field experiments indicate that SOCs and/or COCs in Onondaga Lake inhibit macrophyte growth and survival, limiting colonization and spread of macrophytes in Onondaga Lake as compared to a reference lake (i.e., Otisco Lake). The effects of the ionic waste discharged into Onondaga Lake, including increased salinity concentrations, reduced water transparency, degraded lake sediments, and oncolite formation, as well as natural processes, such as wave action, have resulted in a depauperate macrophyte community in the lake.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

There are no standards, criteria, and guidance that specifically address risk to macrophytes. New York State has narrative water quality standards (6 NYCRR Part 703.2) which regulate physical parameters and aesthetic conditions that impair the best use of the surface water but may not be physically measurable.

Low dissolved oxygen (DO) levels that occur in the deeper waters of the lake do not occur in the shallower waters (shoreline) of the lake. Therefore, low DO is not considered a major limiting factor to macrophyte growth, which is primarily in shallower shoreline areas of water with adequate oxygen. Visibility and associated light availability are relatively low most of the year, with the exception of after the fall turnover in November, which will restrict the presence of macrophytes at greater depths.

The reduced water transparency due to ionic waste contravenes the narrative water quality standard (Part 703.2) for turbidity (and possibly color). Oncolite formation is evidence of past contravention of the same standards (Part 703.2) for suspended, colloidal, and settleable solids. Any present day degradation of the sediments (e.g., wave action causing excessive resuspension of oncolites such that it affects macrophyte growth) contravenes the narrative standard for settleable solids. Any excessive resuspension of Solvay waste in the water column, such as during storms events, would contravene both the narrative standard for turbidity and suspended/settleable solids.

Summary

Sustainability of an aquatic macrophyte community that can serve as a shelter and food source for local invertebrates, fish, and wildlife was assessed using three lines of evidence, as follows:

- Comparison of the Onondaga Lake macrophyte community to reference location communities.
- Evaluation of growth and survival of macrophytes in Onondaga Lake using field and laboratory studies.
- Qualitative evaluation of narrative water quality standards.

The Onondaga Lake aquatic macrophyte community has been impacted by pollution. The community shows lower diversity than other eutrophic lakes in New York State and growth and survival of individual plants is low. Qualitative evaluation of water quality conditions indicate that current water quality is suboptimal for macrophyte growth. Based on field studies and the literature, one of the major influences resulting in the current poor condition of the macrophyte community is the vast amount of ionic waste that has been discharged into the lake. In addition to increasing salinity to the point where only a small number of plant species with a limited distribution could survive in the lake, the ionic waste discharge also resulted in low visibility and degradation of sediments due to physical changes caused by the input of high concentrations of calcium carbonate. Sediment degradation can exacerbate the natural effects of wave action, increasing the difficulties of colonizing and spreading in an area. The formation of oncolites may also restrict the presence of macrophytes, particularly in areas that are subject to strong wave action.

12.2 Assessment Endpoint: Sustainability of a Phytoplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

Does the Phytoplankton Community Structure Reflect the Influence of COCs/SOCs?

In general, the characteristics of the phytoplankton communities of Onondaga Lake reflect the polluted and eutrophic nature of the lake. Concentrations of nutrients have also influenced both the types of species found in the lake and the densities of those species. The effect of mercury contamination on the phytoplankton community is unknown, but it has been shown to bioaccumulate in phytoplankton and subsequently can be passed on to animals feeding on phytoplankton in Onondaga Lake.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

There are no standards, criteria, and guidance that specifically address risk to phytoplankton. However, the summed concentration of total ammonia and nitrate has continuously exceeded levels associated with limitation of phytoplankton growth. Narrative water quality standards (6 NYCRR Part 703.2) have been exceeded in the lake, specifically those for settleable solids (e.g., calcite), which may physically impact phytoplankton.

Summary

The sustainability of a phytoplankton community that can serve as a food source for local invertebrates, fish, and wildlife was assessed using two lines of evidence, as follows:

- Field observations of the Onondaga Lake phytoplankton community.

- Qualitative evaluation of narrative water quality standards.

The phytoplankton community in Onondaga Lake reflects the polluted and eutrophic nature of the lake. Qualitative evaluation of water quality conditions indicate that current water quality is suboptimal for phytoplankton growth. Mercury has been shown to bioaccumulate in phytoplankton in Onondaga Lake, and other COCs may also bioaccumulate, although no analyses have been performed to date. Although the phytoplankton community has been impacted by lake conditions, it still serves as a food source for local invertebrates, fish, and wildlife, and as such passes bioaccumulative contaminants such as mercury on in the food chain.

12.3 Assessment Endpoint: Sustainability of a Zooplankton Community That Can Serve as a Food Source for Local Invertebrates, Fish, and Wildlife

Does the Zooplankton Community Structure Reflect the Influence of COCs/SOCs?

The zooplankton community of Onondaga Lake has been affected by stressors, including salinity and calcium carbonate deposition. Native species of daphnids were replaced by exotic high-salinity-tolerant species during the peak industrial pollution period from the 1950s to the 1980s, and did not return until levels of salinity declined in the late 1980s. Despite recent increases in zooplankton diversity, the zooplankton assemblage of the lake remains depauperate compared to other lakes in the region. High concentrations of mercury in the sediments have been shown to be associated with low hatching success of daphnid eggs in laboratory monitoring. The effect of mercury contamination on juvenile or adult daphnids has not been examined, but mercury has been shown to bioaccumulate in zooplankton in Onondaga Lake and subsequently would be passed on to animals feeding on zooplankton in Onondaga Lake.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Selected COCs detected by Honeywell in lake and tributary surface water in 1992 and 1999 were compared to NYSDEC and USEPA water quality standards, criteria, and guidance. The frequency and magnitude of exceedances in Onondaga Lake and tributary water varied by contaminant, year, location, and depth. With the exception of mercury, all COCs (i.e., barium, copper, lead, manganese, zinc, chlorobenzene, dichlorobenzenes, trichlorobenzenes, and bis[2-ethylhexyl]phthalate) exceeded USEPA chronic aquatic or Tier II water quality values. Mercury concentrations, ranging from 0.0009 to 0.307 µg/L (307 ng/L), exceeded the NYSDEC wildlife value of 0.0026 µg/L at various locations throughout the lake and in the tributaries, but not the chronic water quality value (0.77 µg/L) for the protection of aquatic organisms. Sixty out of 114 samples (53 percent) analyzed for mercury in 1992 and 12 out of 56 samples (21 percent) analyzed for mercury in 1999 had concentrations above the NYSDEC wildlife value.

Other exceedances of surface water standards, criteria, and guidance for the protection of aquatic organisms are as follows:

- Copper exceedances of the NYSDEC and USEPA chronic aquatic water quality value of 11.6 µg/L, the NYSDEC and USEPA acute aquatic water quality value of 17.8 µg/L, and the USEPA Ambient Water Quality Criterion Final Chronic Value (AWQC/FCV) of 14.7 µg/L occurred in tributary surface water, which was only sampled in 1992.

- Lead exceedances of the USEPA chronic water quality value of 3.5 µg/L, the USEPA AWQC/FCV of 3.7 µg/L, and NYSDEC chronic aquatic water quality value of 5.2 µg/L occurred in tributary surface water, which was sampled in 1992.

- Manganese concentrations exceeded the USEPA Tier II aquatic life standard of 880 µg/L in the lake in both 1992 and 1999. However, concentrations were approximately within a factor of two of background levels (using Otisco Lake as a reference lake), which also exceeded the USEPA Tier II aquatic life standard.

- One lake sample analyzed for zinc in 1992 exceeded the NYSDEC chronic aquatic water quality value of 107 µg/L. The remaining 21 exceedances occurred in tributaries. Zinc was not analyzed in the lake in 1999.

- Most exceedances of the NYSDEC chronic standard of 5 µg/L for chlorobenzene and dichlorobenzenes were in tributaries (mainly the East Flume). There was one exceedance in the lake at the Willis Avenue Lakeshore area. Exceedances were found in 1992 and 1999.

- One sample exceeded the NYSDEC chronic trichlorobenzenes standard of 5 μg/L in the southern basin in 1992. Trichlorobenzenes were not analyzed in 1999.

Based on these results, concentrations of contaminants of Onondaga Lake water affect aquatic organisms living in certain areas of the lake.

Stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, and phosphorus, generally exceeded guidelines or background levels. Although lake salinity has dropped to 1.1 parts per thousand (ppt) (Effler et al., 1996; Onondaga Lake Partnership [OLP], 2002), this value is still an order-of-magnitude greater than the average world river salinity (0.11 ppt) and is several times higher than salinity levels in Otisco Lake (0.25 ppt), whose drainage basin is also within the Limestone Belt of central New York State. These levels of salinity are likely to exclude some species of macrophytes from the lake.

The high total ammonia (ammonia and nitrite) concentrations present in Onondaga Lake are in part a result of loads received by the lake from the Metropolitan Syracuse Sewage Treatment Plant (Metro) (Matthews et al., 2000). Currently, upgrades to Metro are being guided by an Amended Consent Judgment (ACJ) from 1998. Decreases in total ammonia concentrations have been made, and improved status has been achieved, with respect to ammonia toxicity standards in the last several years and further reductions are planned through December 2012 (Matthews et al., 2001).

Although concentrations of phosphorus have exceeded the aesthetic effects guidance value, this is considered to have minimal impact on macrophytes in the lake. Nonetheless, under the ACJ, concentrations of total phosphorus will be reduced in two phases over the next ten years. Total phosphorus is to be reduced to 0.12 mg/L by April 2006 and to 0.02 mg/L by December 2012.

The large quantities of ionic waste stressors (e.g., calcium carbonate) deposited on Onondaga Lake sediments are also likely to be detrimental to zooplankton eggs deposited in the sediment. If the disturbance of the sediments in the lake causes resuspension of calcite materials (e.g., oncolites or Solvay waste) such that zooplankton eggs or other aquatic organisms are impacted, then there is a violation of the narrative water quality standards (6 NYCRR Part 703.2).

In summary, concentrations of COCs and SOCs in Onondaga Lake water affect zooplankton living in certain areas of the lake, while the majority of the lake is habitable in terms of chemical water quality. Stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, and phosphorus, exceeded guidelines (when available) or background levels. Although lake salinity has decreased since the closure of the chlor-alkali plant, it likely excludes some species of zooplankton from the lake.

Do Measured Concentrations of COCs/SOCs in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Concentrations of COCs/SOCs in sediments were used as a measurement endpoint to evaluate whether certain zooplankton life stages (e.g., eggs) that spend extended periods in contact with Onondaga Lake sediments could be adversely affected by chemicals and stressors.

Concentrations of COCs in surface sediments exceeded guidelines for all sediment COCs (i.e., arsenic, cadmium, chromium, lead, mercury, nickel, dichlorobenzenes [total], trichlorobenzenes [total], ethylbenzene, toluene, xylenes, hexachlorobenzene, total polycyclic aromatic hydrocarbons [PAHs], phenol, dibenzofurans, chlordanes, heptachlor/heptachlor epoxide, DDT and metabolites, total polychlorinated biphenyls [PCBs], and dioxins/furans).

Exceedances of sediment criteria and/or guidance values for the protection of aquatic organisms are as follows:

- Arsenic was detected in 2000 above the NYSDEC and Ontario Ministry of the Environment (OME) lowest effect level (LEL) of 6 mg/kg, the National Oceanic and Atmospheric Administration (NOAA) effects range-low (ER-L) of 8.2 mg/kg, the USEPA toxic equivalent concentration (TEC) of 12 mg/kg, and the NYSDEC and OME severe effect level (SEL) of 33 mg/kg. Ten out of 19 samples (53 percent) collected in 1992 and 59 out of 85 samples (69 percent) collected in 2000 exceeded the site-specific probable effect concentration (PEC) of 2.4 mg/kg calculated for Onondaga Lake.

- Cadmium was detected in 2000 above the NYSDEC and OME LEL of 0.6 mg/kg, the USEPA TEC of 0.6 mg/kg, the NOAA ER-L of 1.2 mg/kg, the NYSDEC and OME SEL of 10 mg/kg, and the USEPA PEC of 11.7 mg/kg. Forty-five out of 114 samples (39 percent) collected in 1992 and 23 out of 85 samples (27 percent) collected in 2000 exceeded the site-specific PEC of 2.4 mg/kg calculated for Onondaga Lake.

- Chromium was detected in 2000 above the NYSDEC LEL and OME LEL of 26 mg/kg, the USEPA TEC of 56 mg/kg, the NOAA ER-L of 81 mg/kg, the NYSDEC and OME SEL of 110 mg/kg, the USEPA PEC of 159 mg/kg, and the USEPA high no-effect concentration (NEC) of 312 mg/kg. Fifty-four out of 114 samples (47 percent) collected in 1992 and 40 out of 85 samples (47 percent) exceeded the site-specific PEC of 50 mg/kg calculated for Onondaga Lake.

- Lead was detected in 2000 above the NYSDEC and OME LEL of 31 mg/kg, the USEPA TEC of 34 mg/kg, the NOAA ER-L of 47 mg/kg, USEPA NEC of 69 mg/kg, the NYSDEC SEL of 110 mg/kg, the OME SEL of 250 mg/kg, and the

USEPA PEC of 396 mg/kg. Seventy out of 114 samples (61 percent) collected in 1992 and 46 out of 85 samples (54 percent) collected in 2000 exceeded the site-specific PEC of 35 mg/kg calculated for Onondaga Lake.

- Mercury was detected in 2000 above the NYSDEC LEL and NOAA ER-L of 0.15 mg/kg, the OME LEL of 0.2 mg/kg, the NYSDEC SEL of 1.3 mg/kg and the OME SEL of 2 mg/kg. Sixty out of 114 samples (53 percent) collected in 1992 and 86 out of 157 samples collected in 2000 (55 percent) exceeded the site-specific PEC of 2.2 mg/kg calculated for Onondaga Lake.

- Nickel was detected above the NYSDEC and OME LEL of 16 mg/kg, the NOAA ER-L of 21 mg/kg, the USEPA NEC of 38 mg/kg, the USEPA PEC of 39 mg/kg, the USEPA TEC of 40 mg/kg, the NYSDEC SEL of 50 mg/kg, and the OME SEL of 75 mg/kg. Seventy-two out of 114 samples (63 percent) collected in 1992 and 50 out of 85 samples (59 percent) collected in 2000 exceeded the site-specific PEC of 16 mg/kg calculated for Onondaga Lake.

- Dichlorobenzenes (sum) were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 12 μg/gOC and the acute toxicity criterion of 120 μg/gOC. Seventeen out of 114 samples (12 percent) collected in 1992 and 34 out of 85 (40 percent) collected in 2000 exceeded the site-specific PEC of 239 μg/kg calculated for Onondaga Lake.

- Trichlorobenzenes (sum) were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 91 μg/gOC and the acute toxicity criterion of 910 μg/gOC. Three out of 114 samples (3 percent) collected in 1992 and 5 out of 85 samples (6 percent) exceeded the site-specific PEC of 347 μg/kg calculated for Onondaga Lake.

- Ethylbenzene was detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 24 μg/gOC, the NYSDEC acute toxicity criterion of 212 μg/gOC, the USEPA sediment quality benchmark (SQB) of 360 μg/gOC, and the ORNL secondary chronic criterion of 8.9 μg/gOC. One out of 114 samples (< 1 percent) collected in 1992 and 26 out of 61 samples (42 percent) collected in 2000 exceeded the site-specific PEC of 176 μg/kg calculated for Onondaga Lake.

- Toluene was detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 49 μg/gOC, the acute toxicity criterion of 235 μg/gOC, the USEPA SQB of 67 μg/gOC, and the ORNL secondary chronic criterion of 5 μg/gOC. Seventeen out of 114 samples (15 percent) collected in 1992 and 26 out of 62 samples (42 percent) collected in 2000 exceeded the site-specific PEC of 42 μg/kg calculated for Onondaga Lake.

- Xylenes (sum) were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 92 μg/gOC, the acute toxicity criterion of 833 μg/gOC, the USEPA SQB of 2.5 μg/gOC, and the ORNL secondary chronic criterion of 16 μg/gOC. Three out of 114 samples (3 percent) collected in 1992 and 18 out of 37 (49 percent) collected in 2000 exceeded the site-specific PEC of 561 μg/kg calculated for Onondaga Lake.

- Hexachlorobenzene was above the NYSDEC wildlife bioaccumulation sediment criterion of 12 μg/gOC, the OME LEL of 2.0 μg/gOC, and the OME SEL of 24 μg/gOC. Twelve out of 89 samples (13 percent) collected in 1992 and 27 out of 85 samples (32 percent) collected in 2000 exceeded the site-specific PEC of 16 μg/kg calculated for Onondaga Lake.

- Total PAHs were detected above the NOAA ER-L of 4,000 μg/kg and numerous criteria for individual PAH compounds. Site-specific PECs were calculated for individual PAH compounds and ranged between 146 μg/kg for benzo(a)pyrene and 1,436 μg/kg for fluoranthene.

- Phenol was detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.5 μg/gOC and the ORNL secondary chronic criterion of 3.1 μg/gOC. No samples collected in 1992 and 11 out of 85 samples (13 percent) collected in 2000 exceeded the site-specific PEC of 45 μg/kg calculated for Onondaga Lake.

- Dibenzofuran was detected above the ORNL secondary chronic criterion of 42 μg/gOC. Two out of 19 samples (11 percent) collected in 1992 and 13 out of 85 samples (15 percent) collected in 2000 exceeded the site-specific PEC of 372 μg/kg calculated for Onondaga Lake.

- Chlordanes (sum) were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.03 μg/gOC and the NYSDEC wildlife bioaccumulation criterion of 0.006. No samples collected in 1992 and 8 out of 84 samples (10 percent) collected in 2000 exceeded the site-specific PEC of 5.1 μg/kg calculated for Onondaga Lake.

- Heptachlor/heptachlor epoxide (sum) were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 0.01 μg/gOC, the acute toxicity criterion of 13 μg/gOC, and the NYSDEC wildlife bioaccumulation criterion of

0.03 μg/gOC. There were not enough data to calculate a site-specific PEC for heptachlor/heptachlor epoxide.

- DDT and metabolites (sum) were detected above the NYSDEC 4-4'-DDT benthic aquatic life chronic toxicity sediment criterion of 1.0 μg/gOC and the OME LEL of 0.8 μg/gOC. One out of 19 samples (<1.0 percent) collected in 1992 and 5 out of 84 samples (6 percent) collected in 2000 exceeded the site-specific PEC of 30 μg/kg calculated for Onondaga Lake.

- Total PCBs were detected above the NYSDEC benthic aquatic life chronic toxicity sediment criterion of 19 μg/gOC, the NYSDEC wildlife bioaccumulation criterion of 1.4 μg/gOC, and the OME LEL of 7 μg/gOC. Fourteen out of 114 samples (12 percent) collected in 1992 and 42 out of 115 samples (37 percent) collected in 2000 exceeded the site-specific PEC of 295 μg/kg calculated for Onondaga Lake.

- Dioxins/furans were detected above the NYSDEC wildlife bioaccumulation criterion of 0.0002 μg/gOC. There were not enough data to calculate a site-specific PEC for dioxins/furans.

Summary

Sustainability of a zooplankton community that can serve as a food source for local invertebrates, fish, and wildlife was assessed using three lines of evidence, as follows:

- Field observations of the Onondaga Lake zooplankton community.

- Comparison of surface water concentrations to water quality standards, criteria, and guidance developed for the protection of aquatic life and qualitative evaluation of narrative standards.

- Comparison of contaminant concentrations in sediment to guidelines.

All three of these lines of evidence indicate that the zooplankton community of Onondaga Lake has been impacted by high levels of COCs and/or SOCs in lake water. In particular, high levels of salinity and mercury have influenced community structure and abundance. Although the zooplankton community has been impacted by lake conditions, it still serves as a food source for local invertebrates, fish, and wildlife, and as such passes bioaccumulative contaminants such as mercury on in the food chain.

12.4 Assessment Endpoint: Sustainability of a Terrestrial Plant Community That Can Serve as a Shelter and Food Source for Local Invertebrates and Wildlife

Does the Terrestrial Plant Community Structure Reflect the Influence of COCs/SOCs?

The terrestrial plant communities found around Onondaga Lake reflect the development that has occurred near the lake over the last two centuries. Only obvious effects, such as the sparse vegetation found on the wastebeds, can be conclusively attributed to activities at Honeywell facilities (i.e., disposal of Solvay and other industrial wastes).

Do Measured Concentrations of COCs/SOCs in Soil Exceed Toxicity Values for Terrestrial Plants?

Arsenic, cadmium, chromium, lead, mercury, nickel, silver, selenium, thallium, vanadium, and zinc exceeded a hazard quotient (HQ) of 1.0 in plants at one or more of the four wetlands and in the dredge spoils area, indicating that potential risks to plants exist at these locations. In particular, surface soil concentrations of chromium and mercury were over an order-of-magnitude greater than benchmark values. Risks from nickel, selenium, vanadium, and zinc (except at Wetland SYW-6) may be due to background concentrations of these inorganic compounds. Potential risks attributed to site contamination are:

- Wetland SYW- 6: chromium, lead, mercury, nickel, thallium, and zinc.
- Wetland SYW-10: chromium, lead, mercury, and thallium.
- Wetland SYW-12: cadmium, chromium, lead, mercury, and silver.
- Wetland SYW-19: cadmium, chromium, lead, and mercury.
- Dredge spoils area: chromium and mercury.

These results suggest the potential for adverse effects on plants via exposure to COCs in soils at all four wetland areas and the dredge spoils area.

Summary

Sustainability of a terrestrial plant community that can serve as a shelter and food source for local invertebrates and wildlife was assessed using two lines of evidence, as follows:

- Field observations of the Onondaga Lake terrestrial plant community.
- Comparison of surface soil concentrations to plant toxicity values.

There was not enough information on the plant community to determine if it had been affected. Comparisons of soil contaminant concentrations to plant toxicity values indicate that high levels of contaminants, in particular chromium and mercury, may adversely affect the plant community and, subsequently, local invertebrates and wildlife that live or forage in local habitats.

12.5 Assessment Endpoint: Sustainability of a Benthic Invertebrate Community That Can Serve as a Food Source for Local Fish and Wildlife

Does the Benthic Invertebrate Community Structure Reflect the Influence of COCs/SOCs?

Many of the benthic invertebrates communities living in the littoral zone (less than 5 m depth) in Onondaga Lake and the mouths of its tributaries have been impacted to some degree by COCs and/or SOCs. The majority of moderately and severely impacted stations were located between Tributary 5A and Ley Creek, with the most severely impacted stations located between Tributary 5A and Onondaga Creek. Most stations in this area have three metrics of the five metrics that are significantly different than Otisco Lake, which was used as a reference station.

Do Concentrations of Contaminants and Stressors in Sediment Influence Mortality, Growth, or Fecundity of Invertebrates Living In or On Lake Sediments?

The 10-day toxicity tests conducted in 1992 indicated that amphipod toxicity due to the high levels of COCs was confined to an area in the southwestern corner of the lake, along Wastebeds 1 through 8 and along the lakeshore area near Harbor Brook and the East Flume. Most chironomid toxicity was confined to the southern half of the lake in three general areas: 1) off Tributary 5A; 2) Ley Creek; and 3) in the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume).

The results of the 42-day chronic sediment toxicity tests conducted in 2000 showed amphipod toxicity to the high levels of COCs from Tributary 5A to the East Flume and near the Metro outfall. Chironomid toxicity occurred in three areas: 1) from Tributary 5A to the East Flume; 2) off Ninemile Creek; and 3) off Ley Creek.

The results of the sediment toxicity tests confirmed that some Onondaga Lake sediments are toxic to benthic invertebrates and increase mortality and reduce growth and fecundity of these organisms. The most toxic sediments are found in the nearshore zone in the southern part of the lake between Tributary 5A and Ley Creek.

Do Measured Concentrations of Contaminants and Stressors in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Measured concentrations of barium, copper, lead, manganese, mercury, zinc, cyanide, chlorobenzene, dichlorobenzenes, and trichlorobenzenes exceed surface water criteria. Concentrations of contaminants in Onondaga Lake water may affect organisms living in Onondaga Lake, particularly in the southern basin. There were exceedances of surface water criteria in the tributaries emptying in to Onondaga Lake. Macroinvertebrates living at tributary mouths are likely affected by contaminants found in those waters.

Stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, and phosphorus, generally exceeded guidelines (when available) or background levels. The DO in the deeper Onondaga Lake water (greater than 3 m) is often lower than the NYSDEC standard, making the deeper part of the lake uninhabitable by benthic invertebrates. In the hypolimnion, concentrations of sulfide, DO, and ammonia currently result in limited use of this portion of the lake by fish and macroinvertebrates.

The large quantities of ionic waste stressors (e.g., calcium carbonate) deposited on Onondaga Lake sediments are also likely to be detrimental to macroinvertebrate eggs deposited in the sediment. If the disturbance of the sediments in the lake causes resuspension of calcite materials (e.g., oncolites or Solvay waste) such that macroinvertebrate eggs or other aquatic organisms are impacted, then there is a violation of the narrative water quality standards (6 NYCRR Part 703.2).

Do Measured Concentrations of Contaminants and Stressors in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Measured surface sediment concentrations of COCs exceed the consensus probable effect concentration (PEC) values at many locations throughout Onondaga Lake. Only nine of the 114 locations sampled in 1992 and five of the 84 locations sampled in 2000 do not have at least one compound exceeding an HQ of 1.0 (i.e., sediment concentration less than the PEC). Many of the ratios of measured sediment concentrations to PECs exceed 10, or even 100, between Tributary 5A and Ley Creek. In addition, these sediment locations have the highest number of compounds – between 11 and over 30 compounds per sample (note that PAH compounds are considered individually) – exceeding their PECs in a sample.

Mercury exceeded its PEC value of 2.2 mg/kg in 49 percent of the 0 to15 cm sediment samples from Onondaga Lake. Samples were collected principally from the East Flume to Harbor Brook, in the central basins of Onondaga Lake, and off Ninemile Creek. BTEX and chlorinated benzenes were found to exceed one or more PEC value for each group of compounds in 28 and 21 percent of the analyzed surface sediment samples, respectively. These exceedances were found to occur primarily from the Interstate 690 (I-690) lakeshore area to Harbor Brook. Total PCBs were found to exceed the PEC value in 20 percent of the collected 0 to 15 cm sediment samples primarily located in the I-690 lakeshore area to Harbor Brook, and off Ley Creek. One or more PAH exceeded its PEC in 53 percent of the analyzed surface sediment samples. Exceedances occurred primarily from Tributary 5A to Ley Creek and off Ninemile Creek.

In addition to the site-specific consensus PECs developed for this BERA, concentrations of COCs in sediments exceeded NYSDEC sediment quality guidelines for all sediment COCs (i.e., arsenic, cadmium, chromium, lead, mercury, nickel, dichlorobenzenes [total], trichlorobenzenes [total], ethylbenzene, toluene, xylenes, hexachlorobenzene, total PAHs, phenol, dibenzofurans, chlordanes, heptachlor/heptachlor epoxide, DDT and metabolites, total PCBs, and dioxins/furans). In particular, sediment guidelines for the protection of wildlife were exceeded lakewide for PCBs and for dioxins/furans immediately off the I-690 lakeshore area, at the mouth of the East Flume, off the Harbor Brook/Metro area, and at the mouths of Ley Creek and Ninemile Creek.

No guidelines address stressors in sediments, in particular the large quantities of ionic waste stressors (e.g., calcium carbonate and the presence of oncolites) deposited on Onondaga Lake sediments. Based on the BERA analysis it can not be clearly defined if there are potential direct effects of calcium carbonate or oncolites on the benthic community in Onondaga Lake, although the above-stated effects on the macrophyte community could affect the benthic invertebrate community.

Summary

Benthic invertebrate community structure as a food source for local fish and wildlife was assessed using four lines of evidence, as follows:

- Evaluation of the benthic invertebrate community structure and abundance relative to regional conditions.
- Examination of the toxicity of sediments collected from different lake locations.
- Comparison of measured water column concentrations to water quality standards, criteria, and guidance for the protection of aquatic life and qualitative evaluation of narrative standards.
- Comparison of measured sediment concentrations to site-specific sediment effect levels developed specifically for the protection of the benthic invertebrate community in Onondaga Lake and to NYSDEC screening criteria.

All four lines of evidence suggest an adverse effect of COCs on the benthic invertebrate populations in Onondaga Lake. Therefore, local fish and wildlife populations using the benthic invertebrate community as a food source are likely in turn to be impacted.

12.6 Assessment Endpoint: Sustainability of Local Fish Populations

What Does the Fish Community Structure Suggest about the Health of Local Fish Populations?

The current level of fish species diversity in Onondaga Lake is similar to values found in other New York State lakes, although species diversity was lower during the time when the chlor-alkali plants operated. In contrast to comparison lakes, the majority of the species found in Onondaga Lake do not reproduce there and recruitment rates are unknown. Many areas of Onondaga Lake are not suitable for fish reproduction due to industrial and municipal pollution and its effects on the lake ecosystem, such as reduced macrophyte cover and depleted DO levels.

The composition of the fish community in the lake varies seasonally, with migration between the Seneca River and the lake being an important contributor to the variability. Several species of fish found in Onondaga Lake generally retreat to deeper cooler waters during hot weather. These are to a great extent the same fish species that migrate out of the lake. When they are unable to use the deeper part of the lake due to low DO, these species can move out of the lake to avoid the heat and low DO, particularly in late summer and early fall. The limited fish reproduction in the lake and migration out of the lake during the fall indicate that Onondaga Lake alone cannot support the full diversity of the current fish community. Only with immigration into Onondaga Lake and refugia used during times of stress is the current diversity of the fish community sustainable.

Has the Presence of COCs/SOCs Influenced Fish Foraging or Nesting Activities?

Fish reproduction within the lake varies by location. Based on the absence of juveniles in the catches of shoreline seine hauls, it is doubtful that some of the larger species such as the walleye (*Stizostedion vitreum*) and northern pike (*Esox lucius*) reproduce in the lake. Based on historical data, the lack of current lack of nursery area and adequate spawning sites has reduced successful reproduction of fish, resulting in poor year classes. Decreased water clarity, calcium carbonate precipitation, and increased salinity have reduced littoral zone vegetation, a critical area for spawning and young-of-year (YOY) fish. Areas characterized by the presence of aquatic macrophytes and submerged structures (e.g., near the lake outlet) supported the largest populations of juveniles. Areas with heavy silt loads and that are unprotected from wind are undesirable as spawning areas, as silt loads or wave action may cause eggs to be covered or removed from optimal areas.

Stressors, such as calcite and high salinity, have altered the phytoplankton and zooplankton communities in the lake, which will in turn impact the food supply of many fish species. The low amount of littoral zone vegetation also results in lower biomass of macroinvertebrates, which serve as primary food for many YOY fish. The conditions in Onondaga Lake have adversely affected fish reproduction and growth, as evidenced by low reproduction in the lake and fewer YOY fish than observed in lakes with similar characteristics.

Do Fish Found in Onondaga Lake Show Reduced Growth or Increased Incidence of Disease (e.g., Tumors, Lesions) as Compared to Fish from Other Lakes?

Limited data are available regarding the incidence of disease in Onondaga Lake fish. The rate of abnormalities observed in the lake in 1992 and in Geddes Brook and Ninemile Creek in 1998 by Honeywell was inconclusive. Therefore, insufficient data are available to evaluate this endpoint.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Concentrations of COCs/SOCs in surface water exceeded surface water criteria in both Onondaga Lake and its tributaries, with more exceedances of surface water criteria in the Onondaga Lake tributaries than in the lake itself. Stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, and phosphorus, generally exceeded guidelines (when available) or background levels. Although lake salinity showed a decline from the high levels measured prior to closure of the chlor-alkali plants, it is currently about 1.1 ppt, which is still an order-of-magnitude greater than the average world river salinity (0.1 ppt). These levels of salinity are likely to exclude some species of fish from the lake. The summed concentration of total ammonia and nitrate has continuously exceeded standards to protect non-salmonid (as well as salmonid) fish.

The DO in the hypolimnion of Onondaga Lake water generally fails to meet the NYSDEC standard It is likely that in mid- to late summer, when the temperature is reaching its highest level in the epilimnion and DO is reaching its lowest level in the hypolimnion, fish seek deeper, cooler waters. When they are unable to use the deeper part of the lake because of low DO these species move out of the lake to avoid the heat, and possibly other shallow water stressors as well.

The large quantities of ionic waste stressors (e.g., calcium carbonate) deposited on Onondaga Lake sediments are also likely to be detrimental to fish eggs deposited in the sediment. If the disturbance of the sediments in the lake causes resuspension of calcite materials (e.g., oncolites or Solvay waste) such that fish eggs or other aquatic organisms are impacted, then the contaminants in the sediments may cause a violation of the narrative water quality standards (6 NYCRR Part 703.2). Calcite deposition and resuspension, ionic stratification, and oncolite formation have negative effects on the fish community and thereby impair the best use of the water (fish propagation and survival). Water quality standards for turbidity and suspended or settleable solids are exceeded.

Do Measured Concentrations of COCs/SOCs in Sediments Exceed Criteria and/or Guidelines for the Protection of Aquatic Organisms?

Concentrations of COCs in Onondaga Lake sediments exceeded NYSDEC sediment screening criteria for all sediment COCs (i.e., arsenic, cadmium, chromium, lead, mercury, nickel, dichlorobenzenes [total], trichlorobenzenes [total], ethylbenzene, toluene, xylene, hexachlorobenzene, total PAHs, phenol, dibenzofurans, chlordane, heptachlor/heptachlor epoxide, DDT and metabolites, total PCBs, and dioxins/furans). Concentrations of COCs in Onondaga Lake sediments exceeded consensus PECs developed for this BERA for one or more contaminant at about 92 percent of all 1992 sampling locations and about 96 percent of all 2000 sampling locations.

Do Measured Concentrations of COCs in Fish Exceed Toxicity Reference Values for Adverse Effects on Fish?

Risks to fish from contaminants were evaluated on a species-specific basis using measured body burdens. Eight fish species were analyzed to represent the Onondaga Lake fish community. A limited number of contaminants (e.g., methylmercury) were analyzed in some species (e.g., gizzard shad [*Dorosoma cepedianum*] and largemouth bass [*Micropterus salmoides*]); therefore, actual risks from contaminants in lake water may be greater for these species than calculated. HQs greater than 1.0 were calculated for the following contaminants (by fish species):

- Bluegill (*Lepomis macrochirus*) – arsenic, chromium, endrin, mercury, selenium, vanadium, and zinc.

- Carp (*Cyprinus carpio*) – arsenic, chromium, dioxin/furans, endrin, mercury, total PCBs, selenium, vanadium, and zinc.

- Catfish – chromium, endrin, methylmercury, mercury, total PCBs, selenium, vanadium, and zinc.

- Gizzard shad – methylmercury.

- Largemouth bass – methylmercury and dioxins/furans.

- Smallmouth bass (*Micropterus dolomieui*) – arsenic, chromium, mercury, methylmercury, total PCBs, selenium, vanadium, and zinc.

- Walleye – chromium, mercury, methylmercury, and total PCBs.

- White perch (*Morone americana*) – chromium, mercury, methylmercury, selenium, and total PCBs.

Such results suggest adverse effects on most fish species via exposure to COCs in water, sediment, and prey. Concentrations of chromium and mercury exceeded TRVs in all fish species examined (when included in analyses) throughout much of the point estimate range of risk (i.e., from the 95 percent upper confidence limit [UCL] concentration to no observable adverse effect level [NOAEL] HQ to the mean concentration to lowest observable adverse effect level [LOAEL] HQ). Contaminant levels in Onondaga Lake fish were greater than those in background fish. Although background fish HQs for arsenic, chromium, selenium, vanadium, and zinc sometimes exceeded 1.0 (primarily for upper-bound exposure; i.e., the maximum concentration compared to NOAEL), all contaminant concentrations were considered to be site-related. These results indicate that measured concentrations of contaminants in fish are adversely affecting fish populations, particularly where embryos and young are exposed to these levels.

Summary

The sustainability of local fish populations was assessed using six lines of evidence, as follows:

- Examination of the fish community structure as compared to similar lakes and historic accounts of Onondaga Lake (prior to industrial activities) in relation to local fish populations.

- Looking for potential effects of COCs/SOCs on fish foraging and nesting.

- Comparison of visual abnormalities (e.g., tumors, lesions) in Onondaga Lake fish to fish from other lakes.

- Comparison of measured water column concentrations to water quality criteria for the protection of aquatic life and qualitative evaluation of narrative standards.

- Comparison of measured sediment concentrations to guidelines for the protection of aquatic life.

- Comparison of measured concentrations of contaminants in fish representing various feeding strategies and trophic levels to TRVs.

Five of the six lines of evidence suggest adverse effects of COCs on the Onondaga Lake fish community and the remaining line of evidence, incidence of visual abnormalities, was inconclusive. This strength-of-evidence approach indicates that local fish populations are adversely affected by the contaminants and stressors present in Onondaga Lake.

12.7 Assessment Endpoint: Sustainability of Local Amphibian and Reptile Populations

What Do the Available Field-Based Observations Suggest about the Health of Local Amphibian and Reptile Populations?

A field survey of Onondaga Lake concluded that the herpetofauna around the lake was generally depauperate, including the absence of some common species. The number of amphibian and reptilian species found around the lake was considerably lower than the number of species recorded for Onondaga County as a whole. Nearly all amphibians and reptiles appear to inhabit only terrestrial and wetland areas isolated from direct contact with the lake, primarily along the northwest shoreline of the lake.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Aquatic Organisms?

Concentrations of COCs and SOCs in Onondaga Lake water affect organisms living in certain areas of the lake. There were more exceedances of surface water criteria in the Onondaga Lake tributaries than in the lake itself. The majority of wetland areas, some of which could provide suitable habitats for amphibians and reptiles, are found near tributary mouths (e.g., Ninemile Creek, Harbor Brook, and Ley Creek).

Stressors in Onondaga Lake, including chloride, salinity, ammonia, nitrite, sulfide, and phosphorus, generally exceeded guidelines (when available) or background levels. Although lake salinity showed a decline from the highest levels measured prior to closure of the chlor-alkali plants, it is still an order-of-magnitude greater than the average world river salinity (0.1 ppt). These high levels of salinity are detrimental to amphibian reproduction and growth.

Have Laboratory Studies Indicated the Potential for Adverse Effects to Amphibian Embryos from Exposure to Onondaga Lake Water?

The toxicity of water from Onondaga Lake and associated wetlands on developing amphibian embryos was evaluated in laboratory studies (Ducey et al., 2000) and found to have variable, but consistently negative, effects on amphibian development relative to controls.

Summary

Sustainability of local amphibian and reptile populations was assessed using three lines of evidence, as follows:

- Conducting a field survey of local amphibian and reptile populations around Onondaga Lake.
- Comparison of measured water column concentrations to water quality criteria for the protection of aquatic life.
- Performing laboratory studies examining the effects of Onondaga Lake water on amphibian embryos.

All three lines of evidence strongly indicate that amphibian and reptile populations have been adversely affected by contaminants and/or stressors found in Onondaga Lake water.

12.8 Assessment Endpoint: Sustainability of Local Insectivorous Bird Populations

Do Modeled Dietary Doses to Insectivorous Birds Exceed TRVs for Adverse Reproductive Effects?

The tree swallow (*Tachycineta bicolor*) was selected to represent insectivorous birds around Onondaga Lake and was used for modeling dose exposures. Modeled dose concentrations of barium, chromium, methylmercury, mercury, selenium, and total PAHs for the tree swallow exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean concentrations. Cadmium, lead, zinc, dichlorobenzene, total PCBs, and dioxin/furan (toxicity equivalent [TEQ]) dose concentrations exceeded the NOAEL at the 95 percent UCL and mean concentrations. Only selenium and zinc risks may be due to background levels, rather than site contamination. These results suggest adverse effects on insectivorous birds via exposure to COCs in water and particularly in prey.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC wildlife protection value of 0.0026 μg/L. The other contaminants for which there were available wildlife water quality standards, criteria, or guidance values were not detected (i.e., DDT and PCBs) or not analyzed (i.e., dioxins/furans).

What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Bird Populations?

Insectivorous birds have been sighted year-round around Onondaga Lake. However, no site-specific data have been collected; therefore, the significance of bird sightings is uncertain.

Summary

Sustainability of local insectivorous bird populations was assessed using three lines of evidence, as follows:

- Modeling of dietary doses of contaminants.
- Comparison of measured water column concentrations to water quality standards, criteria, and guidance for the protection of wildlife.
- Field-based observation.

The first two lines of evidence indicated that insectivorous birds may be adversely affected by contaminants found in Onondaga Lake and taken up by aquatic invertebrates. The third line of evidence, field observations, was inconclusive.

12.9 Assessment Endpoint: Sustainability of Local Benthivorous Waterfowl Populations

Do Modeled Dietary Doses to Benthivorous Waterfowl Exceed TRVs for Adverse Reproductive Effects?

The mallard (*Anas platyrhynchos*) was selected to represent benthivorous waterfowl around Onondaga Lake and was used for modeling dose exposures. Modeled dose concentrations of barium, cadmium, chromium, dichlorobenzene, dioxins/furans (TEQ), methylmercury, total PAHs, and zinc exceeded TRVs for the mallard. Only zinc risks may be due to background levels, rather than contamination. HQs were exceeded over the range of point estimates of risk. These results suggest adverse effects on waterfowl via exposure to COCs in water, sediment, and particularly dietary sources.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC wildlife protection value of 0.0026 μg/L. The other contaminants for which there were available wildlife water quality standards, criteria, or guidance values were not detected (i.e., DDT and PCBs) or not analyzed (i.e., dioxins/furans).

What Do the Available Field-Based Observations Suggest About the Health of Local Waterfowl Populations?

Onondaga Lake is a year-round home to many waterfowl. Although it is clear that Onondaga Lake is an important resource for many resident and migratory species of waterfowl, its significance as a breeding area is unknown and little can be inferred about the health of local waterfowl populations.

Summary

Sustainability of local waterfowl populations was assessed using three lines of evidence, as follows:

- Modeling dietary doses of contaminants.
- Comparison of measured water column concentrations to water quality criteria standards, criteria, and guidance for the protection of wildlife.
- Performing field-based observations.

The first two lines of evidence indicated that waterfowl may be adversely affected by contaminants found in Onondaga Lake via exposure to contaminated water and food sources. The third line of evidence, field observations, was inconclusive.

12.10 Assessment Endpoint: Sustainability of Local Piscivorous Bird Populations

Do Modeled Dietary Doses to Piscivorous Birds Exceed TRVs for Adverse Reproductive Effects?

Three piscivorous birds, the belted kingfisher (*Ceryle alcyon*), great blue heron (*Ardea herodias*), and osprey (*Pandion haliaetus*), were selected to represent piscivorous birds around Onondaga Lake. These species represent a range of prey size preferences and feeding methods. The following COCs exceeded a hazard quotient of one in each species:

- Belted kingfisher – methylmercury, total PAHs, DDT and metabolites, total PCBs, and dioxins/furans.

- Great blue heron – methylmercury, total PAHs, DDT and metabolites, total PCBs, and zinc.

- Osprey – methylmercury, DDT and metabolites, total PCBs, and zinc.

For all three piscivorous birds, modeled methylmercury dose exposure concentrations exceeded NOAEL and LOAEL TRVs at both 95 percent UCL and mean concentrations by up to an order-of-magnitude. All fish COCs were considered to be site-specific based upon comparisons to background levels. These results suggest adverse effects on piscivorous birds via exposure to COCs in water, sediment, and particularly dietary sources.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC wildlife protection value of 0.0026 μg/L. The other contaminants for which there were available wildlife water quality standards, criteria, or guidance values were not detected (i.e., DDT and PCBs) or not analyzed (i.e., dioxins/furans).

What Do the Available Field-Based Observations Suggest About the Health of Local Piscivorous Bird Populations?

Onondaga Lake is a year-round home to a number of piscivorous bird species. However, very few piscivorous species are listed as confirmed or probable breeders. The mud flats area around the mouth of Ninemile Creek is considered by local birders to be a sensitive migratory area that provides good habitat for birds, but also has high concentrations of some contaminants. Insufficient field data are available to evaluate the health of local piscivorous bird populations.

Summary

Sustainability of local piscivorous bird populations was assessed using three lines of evidence, as follows:

- Modeling dietary doses of contaminants.

- Comparison of measured water column concentrations to water quality standards, criteria, and guidance for the protection of wildlife.

- Performing field-based observations.

The first two lines of evidence indicated that piscivorous birds may be adversely affected by contaminants found in Onondaga Lake, and in particular by mercury, for which HQs were greater than 1.0 for the full point estimate range of risk for all three species. The third line of evidence, field observations, was inconclusive.

12.11 Assessment Endpoint: Sustainability of Local Carnivorous Bird Populations

Do Modeled Dietary Doses to Carnivorous Birds Exceed TRVs for Adverse Reproductive Effects?

The red-tailed hawk (*Buteo jamaicensis*) was selected to represent carnivorous birds around Onondaga Lake and was used for modeling dose exposures. Modeled PAH exposure dose concentrations exceeded NOAEL and LOAEL TRVs at both 95 percent UCL and mean concentrations. Modeled doses of dioxins/furans (TEQ) exceeded the NOAEL at both the 95 percent UCL and mean concentrations and the DDT NOAEL was exceeded for the 95 percent UCL concentration. No exceedances were attributable to background risks.

These results suggest adverse effects on carnivorous birds via exposure to COCs in water, soil, and dietary sources.

What Do the Available Field-Based Observations Suggest About the Health of Local Carnivorous Bird Populations?

Covertypes around Onondaga Lake may support carnivorous bird species and a number of species have been confirmed to breed around Onondaga Lake. However, populations of carnivorous birds have not been studied at Onondaga Lake to place these observations into the proper perspective and the health of local carnivorous bird populations cannot be evaluated.

Summary

Sustainability of local carnivorous bird populations was assessed using two lines of evidence, as follows:

- Modeling dietary doses of contaminants.
- Field-based observations.

Modeled dietary doses indicated that carnivorous birds may be adversely affected by contaminants found in Onondaga Lake, and in particular by total PAHs, for which HQs were greater than 1.0 for the full point estimate range of risk. The second line of evidence, field observations, was inconclusive.

12.12 Assessment Endpoint: Sustainability of Local Insectivorous Mammal Populations

Insectivorous receptors around Onondaga Lake were divided into insectivores feeding on aquatic invertebrates and insectivores feeding on terrestrial invertebrates. The little brown bat (*Myotis lucifugus*) was selected to represent insectivorous mammals feeding on aquatic invertebrates and the short-tailed shrew (*Blarina brevicauda*) was used as the representative receptor for insectivorous mammals feeding on terrestrial prey around Onondaga Lake. These two species were used for modeling dose exposures of insectivorous receptors.

Do Modeled Dietary Doses to Insectivorous Mammals Feeding on Aquatic Invertebrates Exceed TRVs for Adverse Reproductive Effects?

Modeled dose concentrations of barium, chromium, methylmercury, and total PAHs for the little brown bat exceeded TRVs over the full range of risk estimates. Arsenic, cadmium, copper, dioxins/furans (TEQ), hexachlorobenzene, mercury, and vanadium exceeded an HQ of 1.0 over a portion of the range of risk estimates. Risks from arsenic, copper, and vanadium may be attributable to background levels of contaminants.

These results suggest adverse effects on insectivorous mammals via exposure to COCs in water, and in particular to insect prey with an aquatic life-phase.

Do Modeled Dietary Doses to Insectivorous Mammals Feeding on Terrestrial Invertebrates Exceed TRVs for Adverse Reproductive Effects?

Due to the small home range of the short-tailed shrew, the four wetland areas (SYW-6, SYW-10, SYW-12, and SYW-19) and the dredge spoils area were modeled individually. Modeled dietary doses were greater than TRVs for the following contaminants (by area):

- Wetland SYW-19, along the southwest corner of the lake near the mouths of Harbor Brook and the East Flume, had the greatest number of exceedances, with modeled doses of methylmercury, dieldrin, dioxins/furans, hexachlorobenzene, and total PAHs exceeding LOAELs and NOAELs at both the 95 percent UCL and mean concentrations (full range of exposure estimates). NOAELs for arsenic, cadmium, lead, selenium, vanadium, trichlorobenzenes, and total PCBs were exceeded at both upper and mean dose exposures. Most contaminants, with the exception for arsenic, selenium, thallium, and vanadium, were considered to be site-related (i.e., above reference or background levels). Wetland SYW-19 will be evaluated further as part of the Wastebed B/Harbor Brook site RI/FS.

- Wetland SYW-10, on the west side of the lake near the mouth of Ninemile Creek, had ten COCs with HQs over 1.0. Modeled doses of methylmercury and total

PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Arsenic, cadmium, dioxins/furans, hexachlorobenzene, lead, selenium, thallium, and vanadium TRVs were also exceeded at one or more points within the risk estimate range. Arsenic, cadmium, lead, methylmercury, hexachlorobenzene, and dioxins/furans were considered to be site-related. Wetland SYW-10 will be evaluated further as part of the Geddes Brook/Ninemile Creek site RI/FS.

- Wetland SYW-6, on the northwest side of the lake, had ten COCs with HQs over 1.0. Modeled doses of methylmercury and total PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Arsenic, cadmium, chromium, dioxins/furans, lead, total PAHs, thallium, and vanadium HQs were above 1.0. Cadmium, chromium, lead, methylmercury, and dioxins/furans were considered to be site-related.

- Wetland SYW-12, at the southeast end of the lake between Ley and Onondaga Creeks, had eight COCs with HQs over 1.0. Methylmercury and total PAHs exceeded LOAELs and NOAELs at both the 95 percent UCL and mean concentrations. Arsenic, cadmium, dieldrin, hexachlorobenzene, lead, and vanadium HQs were above 1.0. Cadmium, dieldrin, lead, methylmercury, and hexachlorobenzene were considered to be site-related.

- The dredge spoils area surface soils had arsenic, hexachlorobenzene, total PAHs, selenium, and vanadium HQs above 1.0. Hexachlorobenzene was considered to be site-related. The dredge spoils area will be evaluated further as a separate site with its own investigation.

These results suggest adverse effects on insectivorous mammals via exposure to COCs in water, sediment, and primarily terrestrial prey in all four wetland areas. Risks from exposure to surface soils in the dredge spoils area are comparable to those of background locations.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC wildlife protection value of 0.0026 μg/L. The other contaminants for which there were available wildlife water quality standards, criteria, or guidance values were not detected (i.e., DDT and PCBs) or not analyzed (i.e., dioxins/furans).

What Do the Available Field-Based Observations Suggest About the Health of Local Insectivorous Mammal Populations?

It is difficult to separate out the effects of chemical contamination on wildlife from those of habitat loss and development. Insectivorous species, such as shrews and bats, are found in covertypes around the lake, but local populations of insectivorous mammals have not been studied to determine whether they have been impacted.

Summary

Sustainability of local insectivorous mammal populations was assessed using three lines of evidence, as follows:

- Modeling dietary doses of contaminants.
- Comparison of measured water column concentrations to water quality standards, criteria, and guidance for the protection of wildlife.
- Field-based observation.

The first two lines of evidence indicated that insectivorous mammals feeding on aquatic invertebrates and insectivorous mammals feeding on terrestrial invertebrates in the wetlands around Onondaga Lake may be adversely affected by contaminants found in Onondaga Lake and the adjacent wetlands. The third line of evidence, field observations, was inconclusive.

12.13 Assessment Endpoint: Sustainability of Local Piscivorous Mammal Populations

Do Modeled Dietary Doses to Piscivorous Mammals Exceed TRVs for Adverse Effects on Reproduction?

Two piscivorous mammals, the mink (*Mustela vison*) and river otter (*Lutra canadensis*), were selected to represent piscivorous mammals. As these piscivorous mammals show differences in prey selection and composition, use of these two receptors is considered to represent the range of potential piscivorous mammal exposure. The following COCs exceeded an HQ of 1.0 in each species:

- Mink – dioxins/furans, hexachlorobenzene, methylmercury, total PAHs, and total PCBs.
- River otter – DDT and metabolites, dioxins/furans, methylmercury, total PAHs, and total PCBs.

Modeled dose concentrations of total PCBs in the mink exceeded the NOAEL and LOAEL TRVs at both the 95 percent UCL and mean exposure doses. Modeled mink dietary doses of methylmercury, total PAHs, and dioxins/furans exceeded NOAELs at the 95 percent UCL and mean exposure dose levels and the LOAEL at the 95 percent UCL. Modeled dose concentrations of methylmercury and total PCBs in the river otter exceeded NOAEL and LOAEL TRVs at both the 95 percent UCL and mean exposure doses. Modeled river otter dietary doses of total PAHs, DDT and metabolites, and dioxins/furans exceeded NOAELs at the 95 percent UCL and mean exposure dose levels and DDT and metabolites also exceeded the LOAEL at the 95 percent UCL concentration for the river otter. All contaminants in fish prey are considered to be site-related. These results suggest adverse effects on piscivorous mammals via exposure to COCs in water, sediment, and primarily prey and exceedances of HQs over the range of exposure and toxicity concentrations modeled indicate a high likelihood of risk.

Do Measured Concentrations of COCs/SOCs in Surface Water Exceed Standards, Criteria, and Guidance for the Protection of Wildlife?

Mercury concentrations in Onondaga Lake in 1992 and 1999 exceeded the NYSDEC wildlife protection value of 0.0026 μg/L. The other contaminants for which there were available wildlife water quality standards, criteria, or guidance values were not detected (i.e., DDT and PCBs) or not analyzed (i.e., dioxins/furans).

What do the Available Field-Based Observations Suggest About the Health of Local Piscivorous Mammal Populations?

It is difficult to separate out the effects of chemical contamination on wildlife from those of habitat loss and development. The mink and river otter are the only primarily piscivorous mammals found in these covertypes; however, there are no data regarding mink or otter populations in the area of Onondaga Lake. The lack of field observations of mink and few otter observations may indicate depressed populations although both species are secretive and can be difficult to see in the field. Populations of mink and otter could be impacted by inadequate habitat, disturbance by humans, and chemical contamination.

Summary

The sustainability of local piscivorous mammal populations was assessed using three lines of evidence, as follows:

- Modeling dietary doses of contaminants.
- Comparison of measured water column concentrations to water quality standards, criteria, and guidance for the protection of wildlife.
- Performing field-based observations.

The first two lines of evidence indicated that piscivorous mammals feeding around Onondaga Lake may be adversely affected by contaminants found in Onondaga Lake, in particular by mercury and total PCBs. The third line of evidence, field observations, was inconclusive.

12.14 Summary

Multiple lines of evidence were used to evaluate major components of the Onondaga Lake ecosystem to determine if lake contamination has adversely affected plants and animals around the lake. Contaminants and stressors in the lake have either impacted or potentially impacted every trophic level and feeding preference examined in this BERA. The aquatic macrophytes in the lake have been adversely affected by lake conditions, and the resulting loss of habitat that formerly provided valuable feeding and nursery areas has undoubtably impacted the aquatic invertebrates and vertebrates living in Onondaga Lake. In addition to general aquatic habitat loss occurring at the time that massive quantities of ionic waste were dumped into the lake, there has been bioaccumulation of mercury and possibly other contaminants in most organisms serving as a food source in the lake, including phytoplankton, zooplankton, benthic invertebrates, and fish.

Almost all lines of evidence indicate that the COCs and ionic waste in Onondaga Lake have produced adverse ecological effects at all trophic levels examined. Comparisons of measured tissue concentrations and modeled doses of contaminants to TRVs show exceedances of HQs throughout the range of the point estimates of risk. Many of the contaminants in the lake are persistent and, therefore, the risks associated with these contaminants are unlikely to decrease significantly in the short-term.

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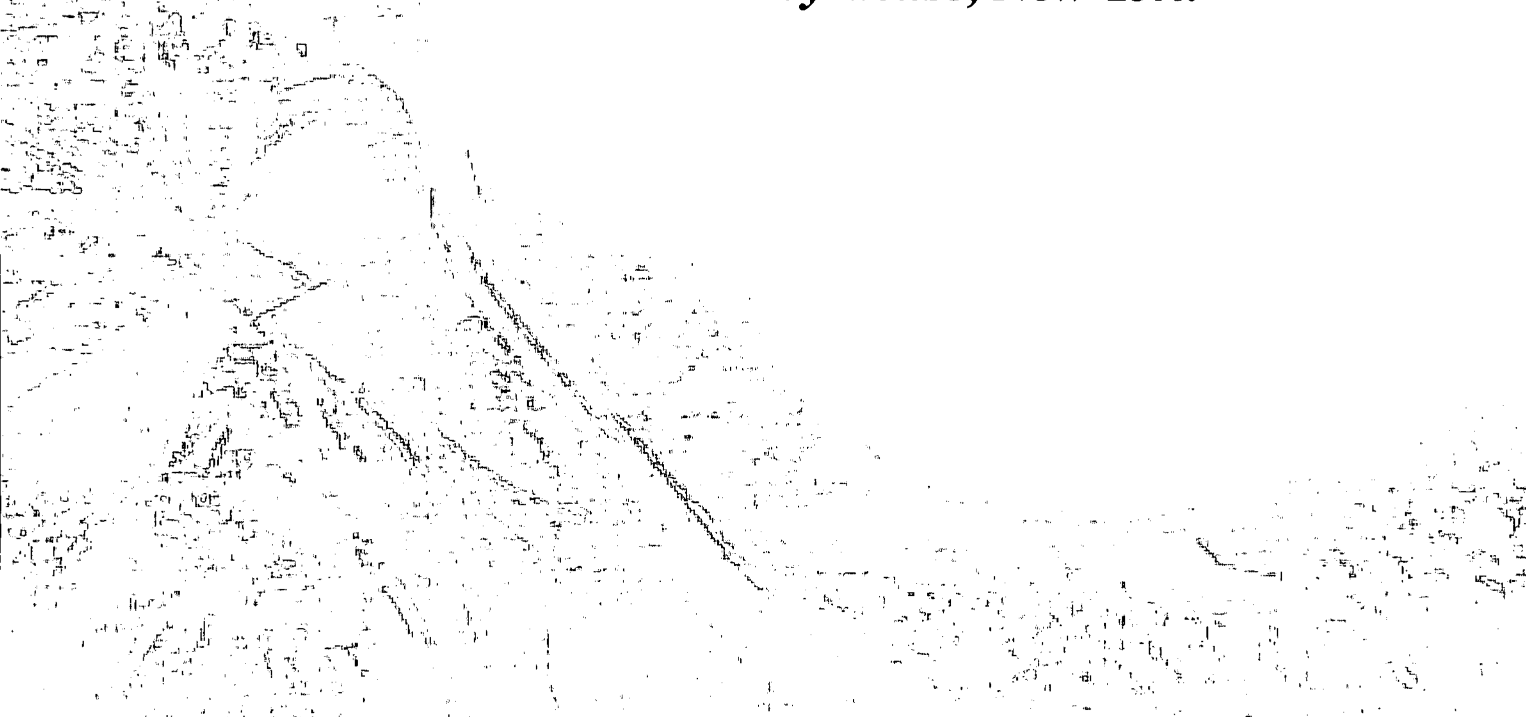

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